Pre-operational Environmental Monitoring Report for the Point Lepreau, N.B., Nuclear Generating Station - 1982

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October 1984

Canadian Technical Report of Hydrography and Ocean Sciences No. 43

Canada

86-03146/

Canadian Technical Report of Hydrography and Ocean Sciences

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Regional and headquarters establishments of Ocean Science and Surveys ceased publication of their various report series as of December 1981. A complete listing of these publications and the last number issued under each title are published in the *Canadian Journal of Fisheries and Aquatic Sciences*, Volume 38: Index to Publications 1981. The current series began with Report Number 1 in January 1982.

Rapport technique canadien sur l'hydrographie et les sciences océaniques

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Les rapports techniques peuvent être considérés comme des publications à part entière. Le titre exact figure au-dessus du résumé du chaque rapport. Les résumés des rapports seront publiés dans la revue Résumés des sciences aquatiques et halieutiques et les titres figureront dans l'index annuel des publications scientifiques et techniques du Ministère.

Les rapports techniques sont produits à l'échelon régional mais sont numérotés et placés dans l'index à l'échelon national. Les demandes de rapports seront satisfaites par l'établissement auteur dont le nom figure sur la couverture et la page de titre. Les rapports épuisés seront fournis contre rétribution par des agents commerciaux.

Les établissements des Sciences et Levés océaniques dans les régions et à l'administration centrale ont cessé de publier leurs diverses séries de rapports depuis décembre 1981. Vous trouverez dans l'index des publications du volume 38 du *Journal canadien des sciences halieutiques et aquatiques*, la liste de ces publications ainsi que le dernier numéro paru dans chaque catégorie. La nouvelle série a commencé avec la publication du Rapport n° 1 en janvier 1982.

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Cat. No. Fs 97-18/43E ISSN 0711-6764

Correct citation for this publication:

Ellis, K., Nelson, R.W.P., and Smith, J.N., 1984. Pre-operational environmental monitoring report for the Point Lepreau, N.B., nuclear generating station - 1982. Can. Tech. Rep. Hydrogr. Ocean Sci. No. 43: vi + 173 p.

ABSTRACT

Ellis, K., Nelson, R.W.P., and Smith, J.N. 1984. Pre-operational environmental monitoring report for the Point Lepreau, N.B., nuclear generating station - 1982. Can. Tech. Rep. Hydrogr. Ocean Sci. No. 43: vi + 173 p.

The Point Lepreau Environmental Monitoring Program (PLEMP) has been established within the Department of Fisheries and Oceans to assess the environmental impact of radioactive, thermal, and chemical releases from the Point Lepreau, N.B., Nuclear Generating Station located on the Bay of Fundy. The final results for the pre-operational phase of the program, extending from 1979 to 1982 when the reactor began operation, are presented. Baseline radioactivity levels, in addition to other oceanographic and environmental parameters, were measured in the major environmental reservoirs such as seawater, sediments, marine flora and fauna, soils, terrestrial and aquatic organisms, and atmospheric gases and particulates. These results have been used to identify transport pathways for radionuclides through environmental phases in the Point Lepreau region and to determine fluxes of radionuclides along specific pathways. Other studies carried out during the pre-operational phase of this program include a surface and bottom drifter release experiment designed to identify water circulation patterns in the Bay of Fundy, marine and terrestrial ecological baseline surveys, and meteorological experiments characterizing potential dispersion pathways for radionuclides introduced into the atmosphere. Measurements conducted during the operational phase of the reactor lifetime (operational since late 1982 and at full power in March 1983) will be compared to the pre-operational conditions in order to establish the environmental effects of the reactor as a function of time. The ultimate goal of this program is to provide government with a sound scientific basis upon which to assess the environmental implications of the operation of nuclear reactors in coastal regions.

RÉSUMÉ

Ellis, K., Nelson, R.W.P., and Smith, J.N. 1984. Pre-operational environmental monitoring report for the Point Lepreau, N.B., nuclear generating station - 1982. Can. Tech. Rep. Hydrogr. Ocean Sci. No. 43: vi + 173 p.

On a créé, au sein du ministère des Pêches et Océans, le Programme de surveillance écologique de Pointe Lepreau afin d'évaluer les répercussions environnementales des émissions chimiques, thermiques et radioactives de la centrale nucléaire de Pointe Lepreau située dans la baie de Fundy, au Nouveau-Brunswick. On présente les résultats finals de la phase du programme précédant la mise en exploitation de la centrale, soit la phase de 1979 à 1982, année où le réacteur est entré en service. Outre d'autres paramètres océanographiques et environnementaux, on a mesuré la radioactivité ambiante dans les principaux réservoirs environnementaux tels que l'eau de mer, les sédiments, la faune et la flore marines, les sols, les organismes aquatiques et terrestres ainsi que les matières particulaires et les gaz atmosphériques. On a utilisé les données ainsi obtenues pour déterminer les circuits de transport des radionucléides dans l'environnement de la région de Pointe Lepreau et pour déterminer les apports de

radionucléides empruntant des circuits de transport donnés. Au nombre des autres études menées pendant la phase du programme précédant la mise en service du réacteur, mentionnons une étude expérimentale de la dérive de fond et de la dérive superficielle conçue pour déterminer les systèmes de circulation de l'eau dans la baie de Fundy, des relevés écologiques des milieux marin et terrestre, ainsi que des expériences météorologiques caractérisant les éventuels circuits de dispersion des radionucléides introduits dans l'atmosphère. On compare les résultats des mesures faites durant la phase d'exploitation du réacteur (entré en service à la fin de 1982 et fonctionnant à plein régime depuis le mois de mars 1983) avec les résultats des mesures faites avant la mise en service, afin de déterminer les effets environnementaux de l'exploitation du réacteur en fonction du temps. Le but ultime de ce programme est de fournir au gouvernement une base de données scientifiques fiables à partir de laquelle on pourra évaluer les effets de l'exploitation de réacteurs nucléaires sur l'écologie des régions côtières.

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1.0 INTRODUCTION

The Point Lepreau Nuclear Generating Station (NGS) located at Point Lepreau, N.B. on the Bay of Fundy (Fig. 1) houses a 660 Mw CANDU reactor. The Point Lepreau NGS attained criticality in July 1982 and gradually increased power to 75% by late December 1982 and to full power by late March 1984, making it the first nuclear reactor operating on the Canadian coastline. As a result, radioactive, thermal and chemical releases are discharged principally into the marine environment, thus presenting potential problems which have not been previously encountered at other nuclear installations in Canada. In particular, speciation and physico-chemical partitioning of radionuclides released into marine systems will differ from that of radionuclides entering freshwater systems. In response to the unique environmental constraints posed by the proximity of this reactor to the ocean, the Point Lepreau Environmental Monitoring Program (PLEMP) has been established within the Department of Fisheries and Oceans to assess the long term environmental impact of the Point Lepreau NGS.

In contrast to most previous environmental monitoring programs for nuclear reactors, which have been primarily concerned with the impact of radioactivity releases on human health, PLEMP has been designed to provide a broader understanding of the distribution of radioactivity throughout the environment. Radioactivity measurements are conducted on samples collected from the major environmental reservoirs such as seawater, sediments, atmospheric gases and particulates and marine flora and fauna. These measurements, in conjunction with the determination of other oceanographic and ecological parameters are being used to identify transport pathways for radionuclides through environmental phases and to determine fluxes of

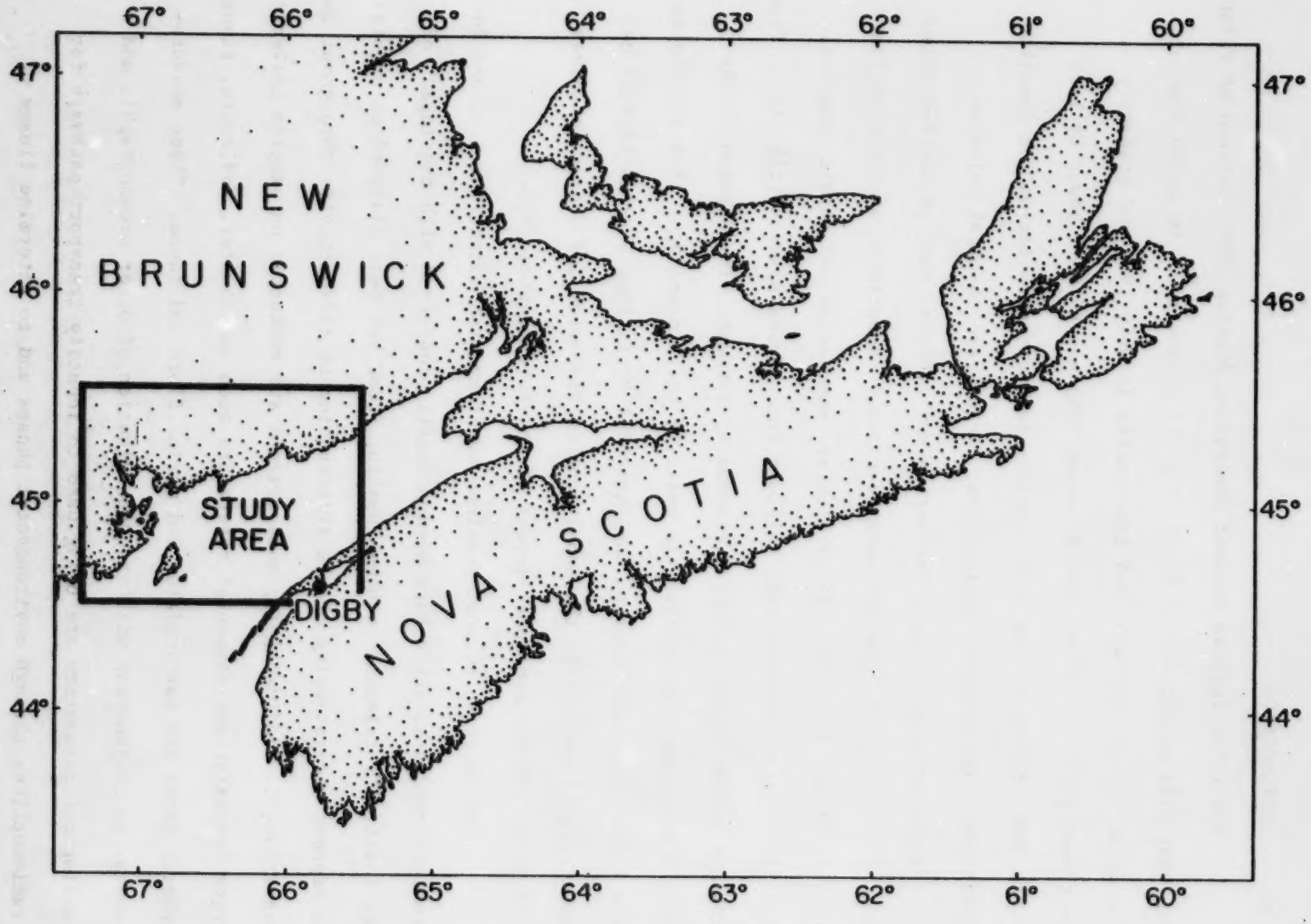


Figure 1. Location of the Point Lepreau study area in the Bay of Fundy.

radionuclides along specific pathways.

The monitoring program is divided into pre-operational and operational phases. The pre-operational phase of PLEMP includes work carried out from 1979 to 1982. During the pre-operational phase of PLEMP, baseline radioactivity and chemical measurements were performed in order to characterize the background radioactivity conditions within the vicinity of Point Lepreau. Measurements conducted during the operational phase of the reactor lifetime (after January, 1984) will be compared to the pre-operational conditions in order to document any environmental effects associated with the operation of the reactor. The ultimate goal of the program is to provide government with a sound scientific basis upon which to assess the environmental implications of the operation of nuclear reactors in coastal environments.

Responsibility for PLEMP has been assigned to the Atlantic Environmental Radioactivity Unit (AERU), which is a section of the Chemical Oceanography Division located at the Bedford Institute of Oceanography, Dartmouth, Nova Scotia. In addition, a working group on environmental radioactivity, composed of scientists in various disciplines and departments in the Atlantic region (Appendix 1), has been established to ensure that the concerns and responsibilities of different agencies with respect to Point Lepreau are addressed in a coordinated manner by PLEMP. The working group advises AERU on field activities, reviews reports produced during this program and members of the working group act as scientific authorities for contracts administered by AERU.

This report, the fourth in an annual series of monitoring reports for the Point Lepreau NGS, describes the work conducted during part of 1981 and 1982 and concludes the pre-operational phase of the program.

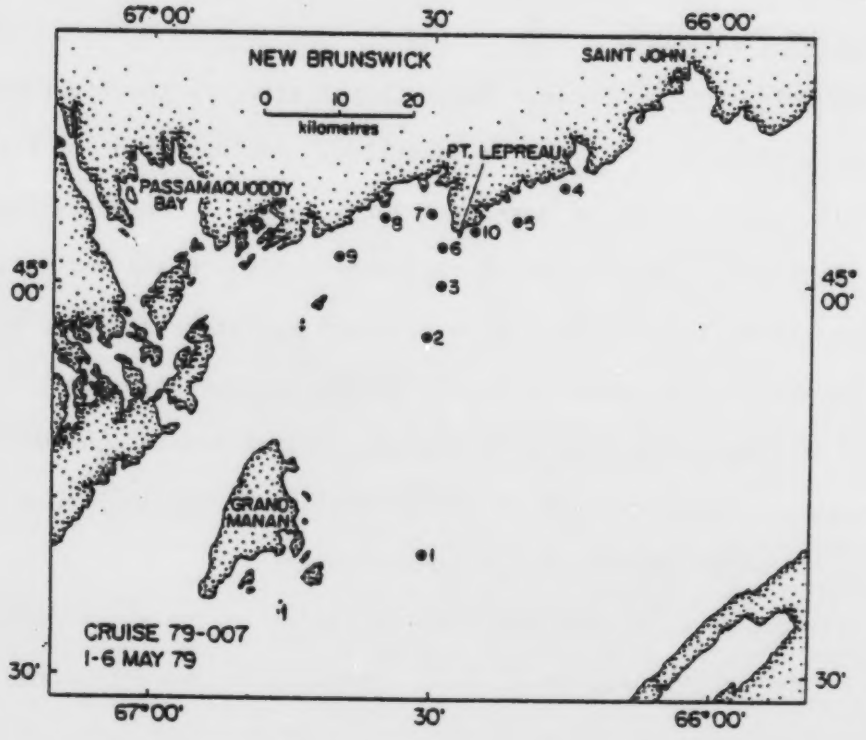
Measurements conducted in 1983 constitute the initiation of the post-operational phase of the program and will be presented in future annual reports.

2.0 MARINE ENVIRONMENT

The Point Lepreau Nuclear Generating Station is the first nuclear reactor to be sited on the Canadian coastline. The operation of this station will result in the release of several types of contamination to the marine environment. The release of radioactive fission and activation products contained in both the liquid effluent and stack gases, are of primary importance. Releases of excess thermal energy with cooling water, discharged at temperatures up to 20°C above ambient levels, and of biocides in the event that they are used to reduce fouling in the cooling system, constitute additional environmental contaminants.

In order to fully characterize the impact of the above releases on the Bay of Fundy ecosystem, it is first necessary to understand the predominate transport processes operating in this region. Initially, it is necessary to characterize the water circulation patterns in the vicinity of Point Lepreau; identify particle transport pathways; establish baseline chemical and radioactivity levels in sea water, suspended matter, sediments and biological phases; and estimate the fluxes of radionuclides through different environmental phases. The 1982 Pre-operational Environmental Monitoring Report for the Point Lepreau NGS summarizes the results for 1982, the final year of the pre-operational program. This completes the pre-operational program begun in 1979 and presents a broad survey of the baseline levels of radioactivity present in the Point Lepreau area.

The bulk of the marine environmental program has been conducted during oceanographic cruises in the Bay of Fundy. There have been five cruises to date in which AERU has participated, two in 1979 (Fig. 2), one in 1980, (Fig. 3) and two in 1981, aboard the CSS Hart (Fig. 3) and the CSS Pandora (April 14-21; Fig. 4). The results of the 1979 cruises and other



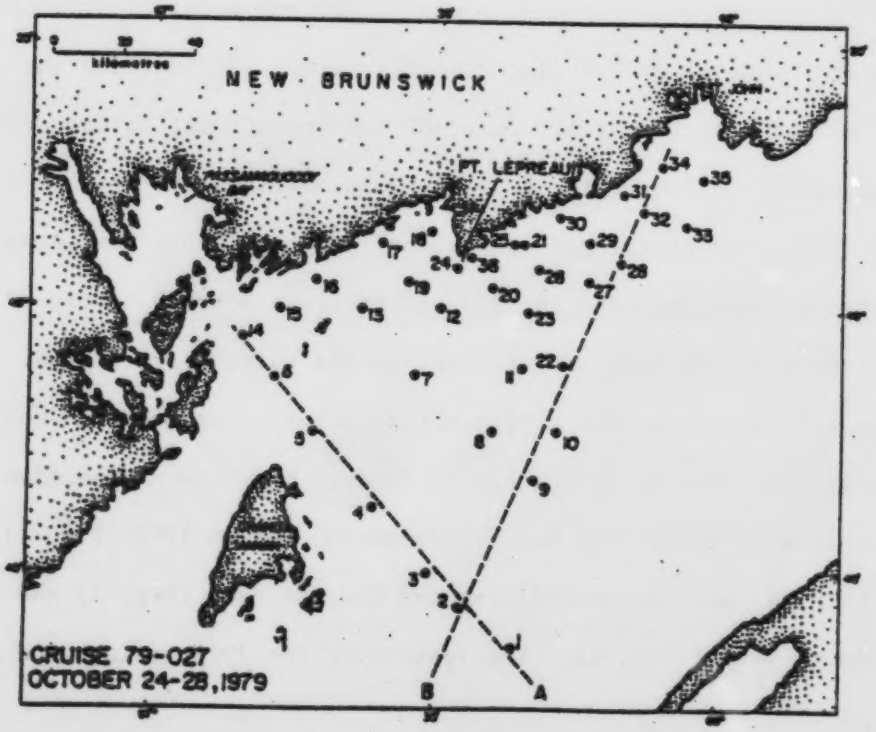
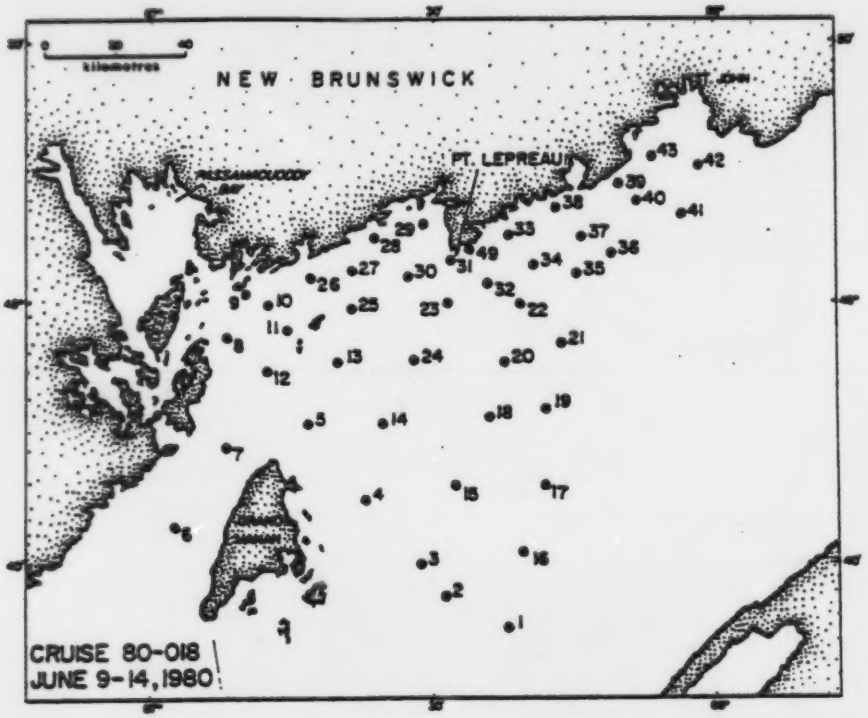


Figure 2. Station locations for cruises 79-007 and 79-027.



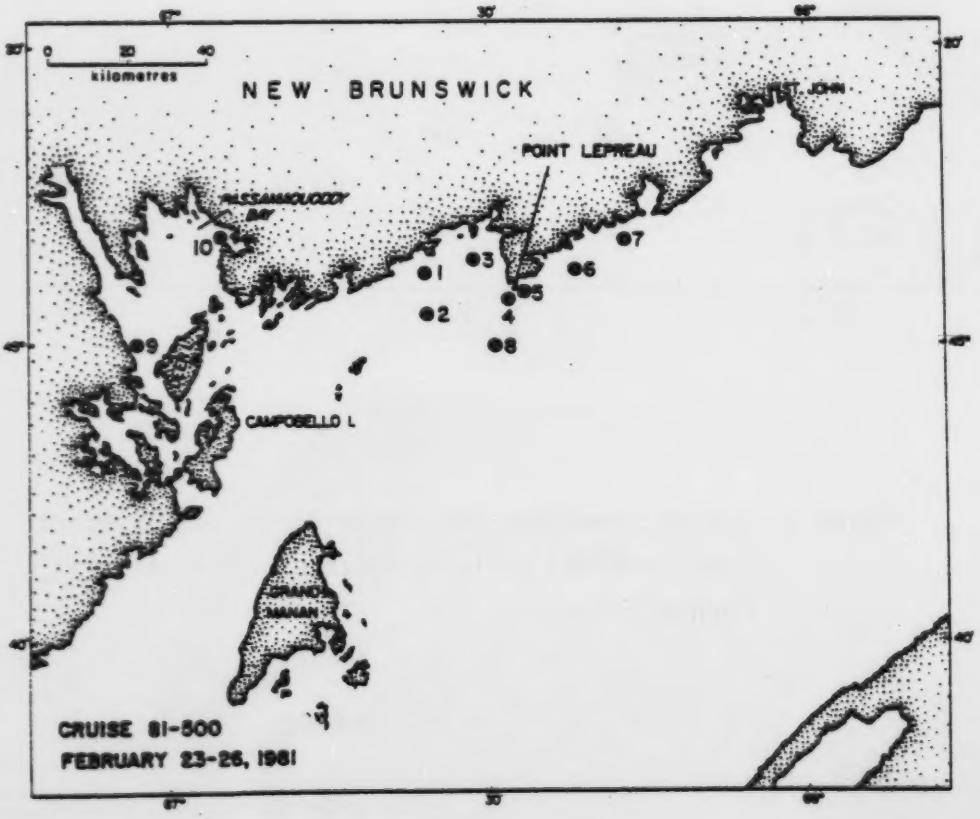


Figure 3. Station locations for cruises 80-018 and 81-500.

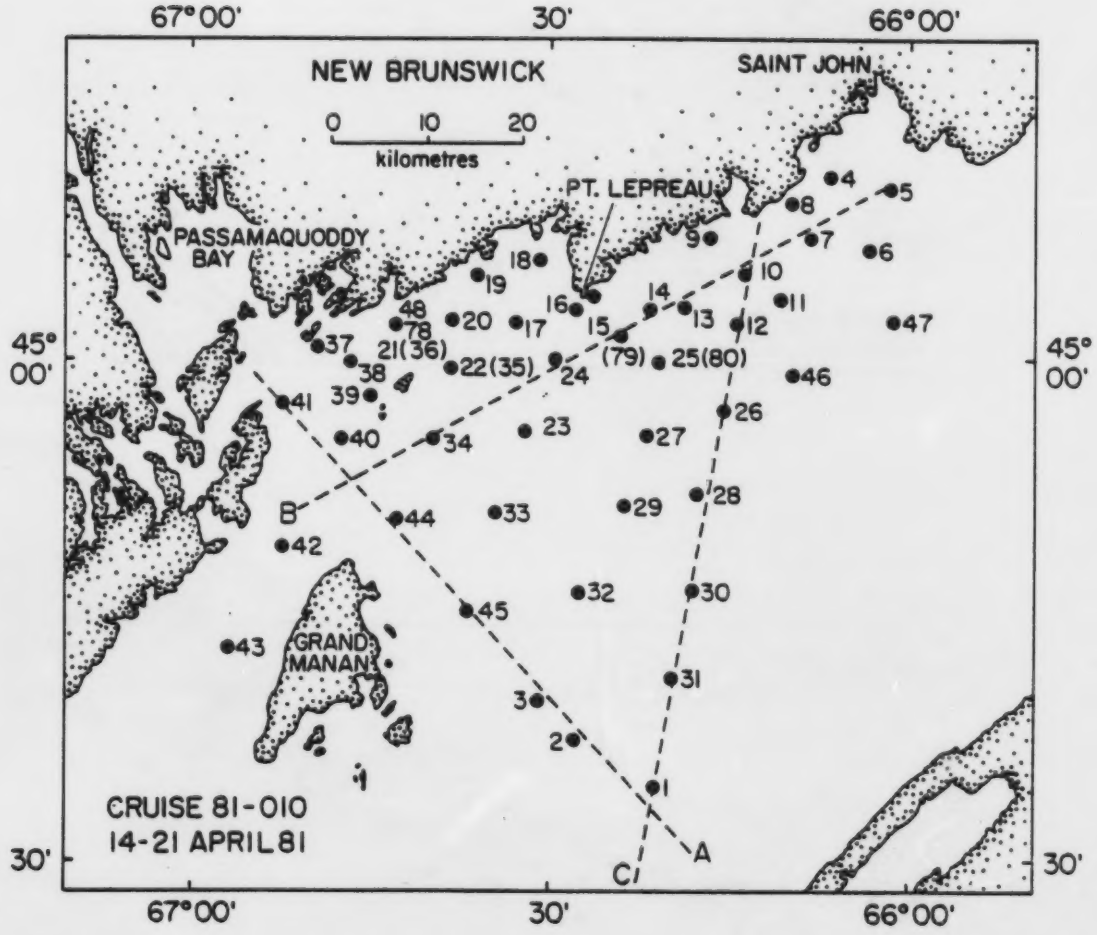
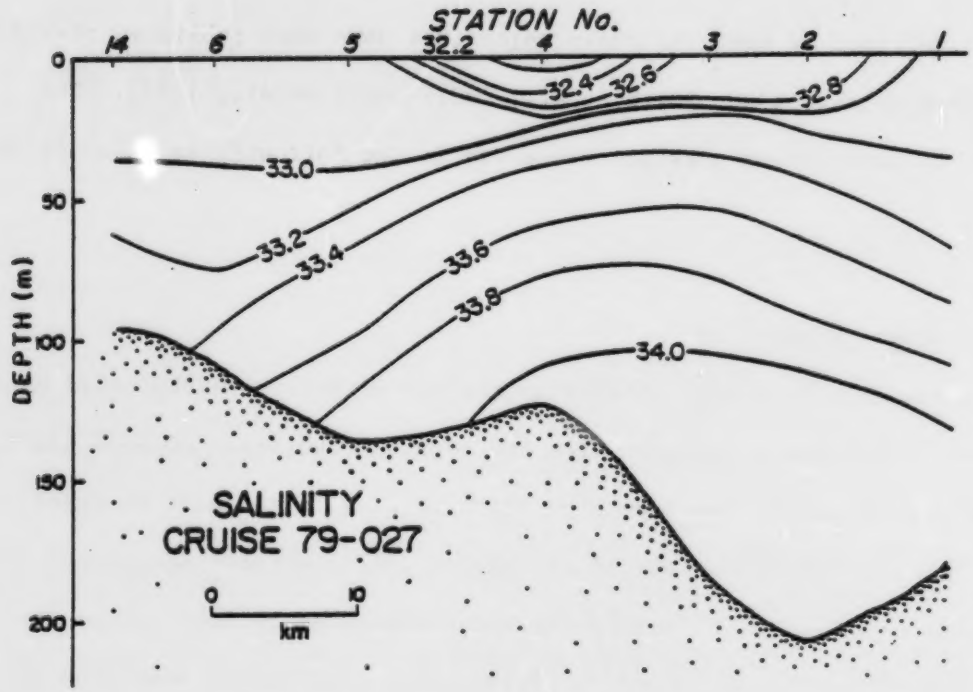


Figure 4. Station locations for cruise 81-010 and transect lines used for salinity-depth profiles shown in Figures 6,7,and 8.

studies designed to meet the above objectives have been published previously (Bishop et al., 1980; Smith et al., 1981; Smith et al., 1982). The results of more recent work are presented in the following sections of this report.

2.1 Water Circulation

Water circulation patterns in the Bay of Fundy are dominated by a cyclonic (anti clockwise) gyre with inflow along the coast of Nova Scotia and outflow along the New Brunswick coast passing to the east of Grand Manan Island. Water movement in the bay is characteristic of simple estuarine circulation. Low salinity water dominated by inputs from the Saint John River (up to 60% of the fresh water input originates from the Saint John River system) flows out at the surface, while denser shelf and slope water mixes across the shelf break and enters the Bay of Fundy at depths predominantly below 100 metres. Salinity-depth profiles have been constructed to help illustrate the predominant circulation patterns in the vicinity of Point Lepreau. Two profiles for the fall cruise 79-027 for transects indicated in Fig. 2 are shown in Fig. 5. Three profiles typical of spring conditions are shown in Figs. 6, 7 and 8 for cruise 81-010 for the transects A,B and C illustrated in Fig. 4. The intrusion at depth of higher salinity (>33°/$_{oo}$) shelf and slope water is evident in both fall and spring (Figs. 5, 6 and 7) where at depths greater than 100 metres salinities are 33°/$_{oo}$ or greater. The freshwater input from the Saint John River is most evident during both spring and fall (transect B, Figs. 5 and 8) but is of much greater magnitude during the high discharge period of the spring. These transects indicate that the plume remains close to the New Brunswick coast until the tip of the Point Lepreau (station 22, cruise



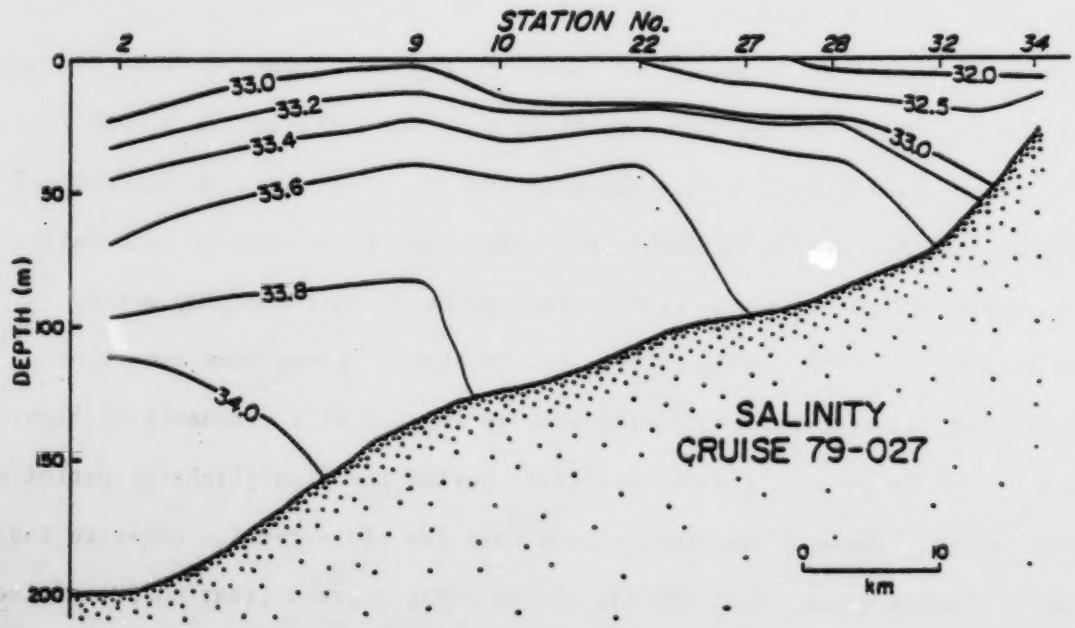


Figure 5. Salinity-depth profiles for transects A and B for cruise 79-027 for transects illustrated in Figure 2.

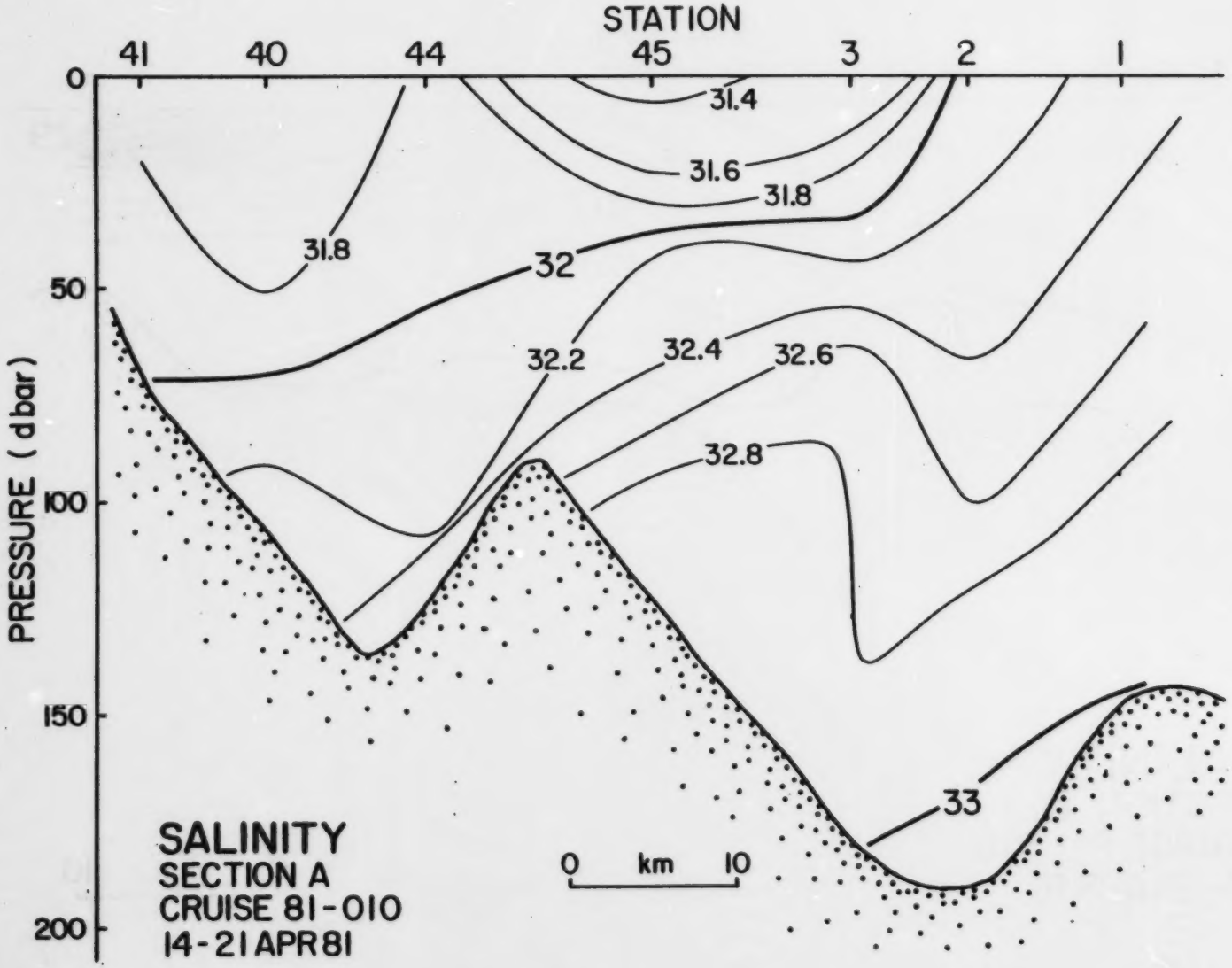


Figure 6. Salinity-depth profile for transect A, Figure 4, Cruise 81-010.

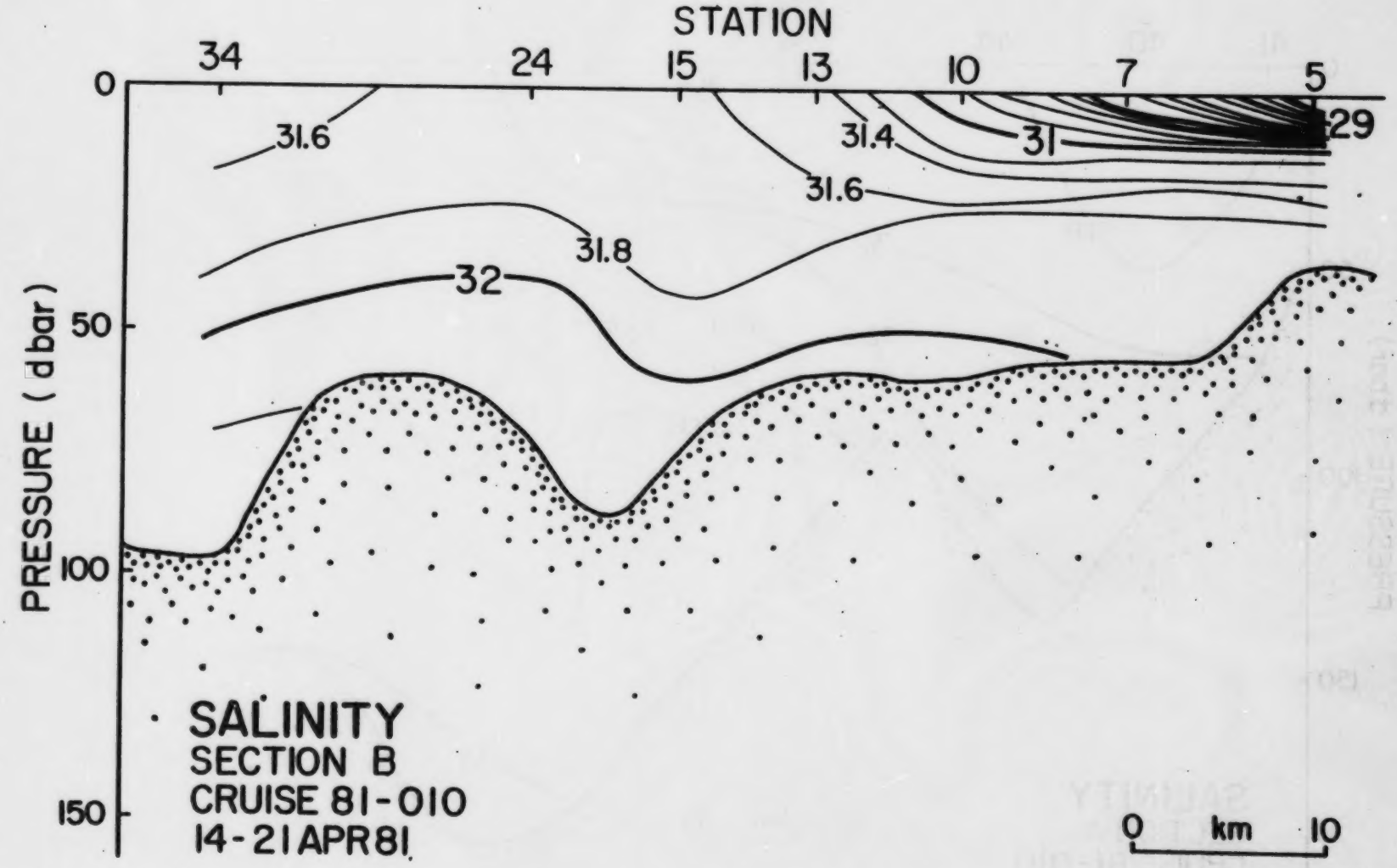


Figure 7. Salinity-depth profile for transect B,Figure 4,Cruise 81-010.

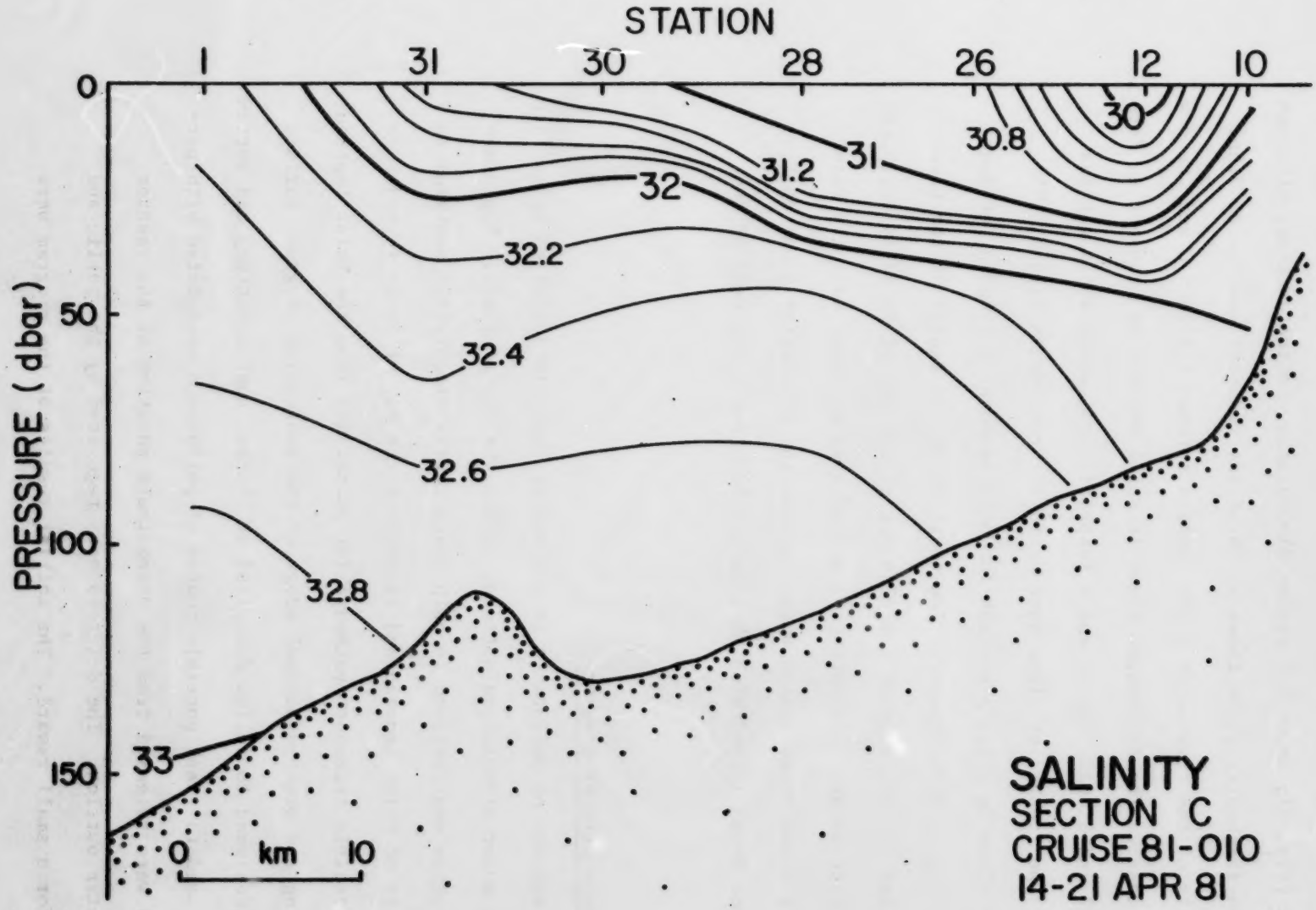


Figure 8. Salinity-depth profile for transect C,Figure 4,Cruise 81-010.

79-027; station 15, cruise 81-010), after which it moves from the shore. Transect C (Fig. 8), shown for cruise 81-010 only, indicates that, although some of the freshwater plume flows south of the transect near Saint John, the majority of the freshwater plume remains between transect C and the New Brunswick Coast. The freshwater plume is still evident in transect A, for-both 79-027 and 81-010 (Fig. 5 and 6 respectively), which shows a cross section of the freshwater plume just north of Grand Manan Island. The freshwater plume is clearly evident for both seasons, but again the magnitude in spring is much greater. The position of the center of the plume suggests that the majority of the flow out of the Bay of Fundy is southeast of Grand Manan Island. In spring, a second area of lower salinity water located off Grand Manan Island suggests that the freshwater plume splits around Grand Manan Island with the smaller fraction going west of the Island.

2.2 Ocean Drifter Program

Transport of pollutants in the Bay of Fundy is dominated by surface and bottom water circulation patterns within the Bay of Fundy. The ocean drifter program was designed to help characterize circulation patterns in the vicinity of Point Lepreau and throughout the Bay of Fundy in order to identify possible transport pathways for pollutants from the Point Lepreau NGS. During the pre-operational stage of the monitoring program, surface drifters (designed to follow dissolved or thermal contamination) and bottom drifters (used to track possible routes of pollutants associated with particulates) were released from the approximate position of the reactor cooling water outflow. The drifters were recovered by the public and returned for a small reward. The initial results of the program were

reported in Bugden (1980) and Smith et al. (1981). The complete program and results are presented in Bugden (1983).

2.2.1 Surface Drifter Returns

The return pattern for surface drifters is shown in Fig. 9. The overall return percentage was 10.9% with return percentages varying from a high of 19.4% in the spring to a low of 7.1% in the winter (Fig. 10). The drifters came ashore in three regions; near the release point and in adjacent Passamaquoddy Bay (60.9%), along the eastern seaboard of the United States (7.6%) and along the northwestern shore of Nova Scotia (31.5%). The surface drifter program reinforced earlier work by Bumpus and Lauzier (1965) and Lauzier (1967); the spring increase in outflow along the northwestern shore of the Bay of Fundy was verified by the seasonal pattern of returns from the U.S. coast. The large number of surface drifters that travelled across the bay to the Nova Scotia shore in late fall and early winter was greater than expected and is indicative of a closed circulation pattern in the Bay of Fundy during winter months.

2.2.2 Bottom Drifter Returns

The pattern observed for bottom drifter returns is shown in Fig. 11. The overall return percentage was 15.3% and the average drift time was 245 days. Fig. 12 categorizes the recovery percentages for various drift times for drifters released during different seasons. The results indicate a pronounced maximum at short recovery times in the spring and winter while fall and summer results are more symmetrical around a longer mean drift time. The similarity in location and seasonal return patterns from year to year for the bottom drifters, despite drift times of more than a year and a

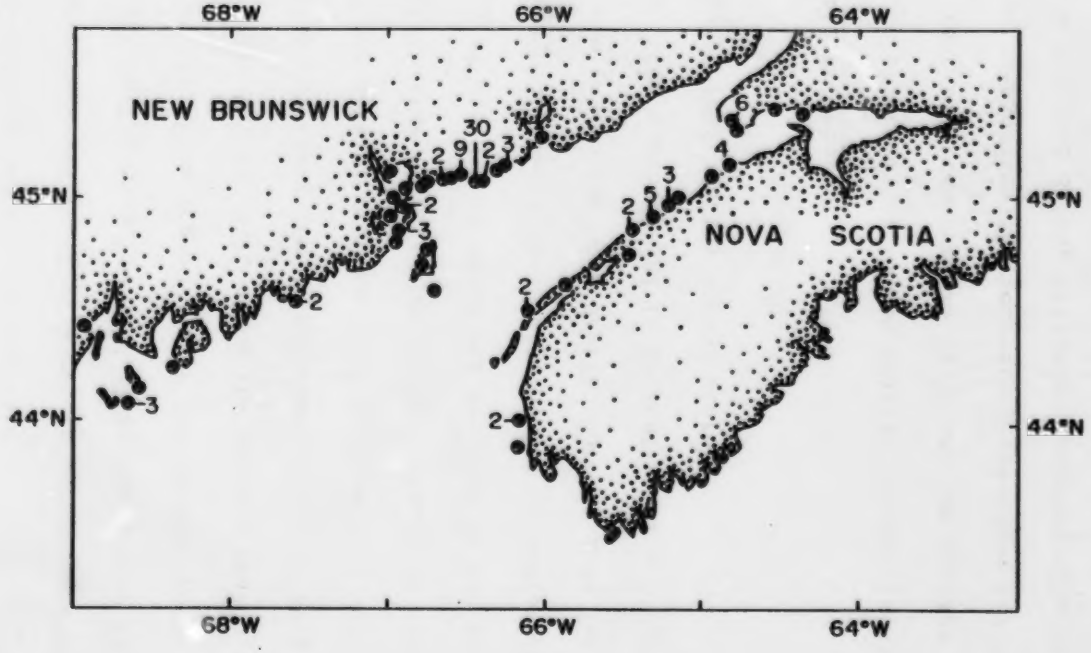


Figure 9. Recovery positions for surface drifter returns. Numbers indicate multiple returns .

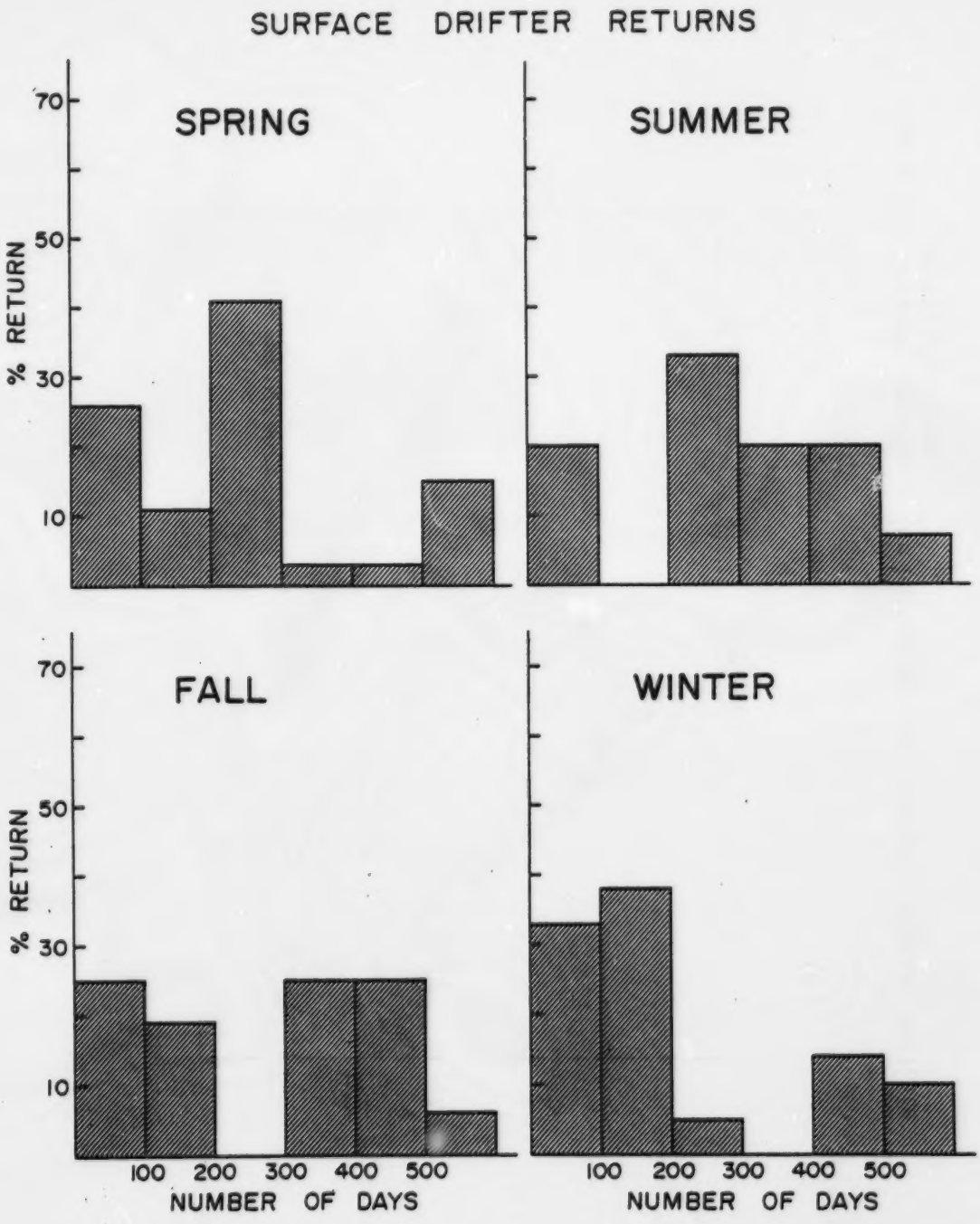


Figure 10. Surface drifter return percentages as a function of drift time, in days, for different seasons.

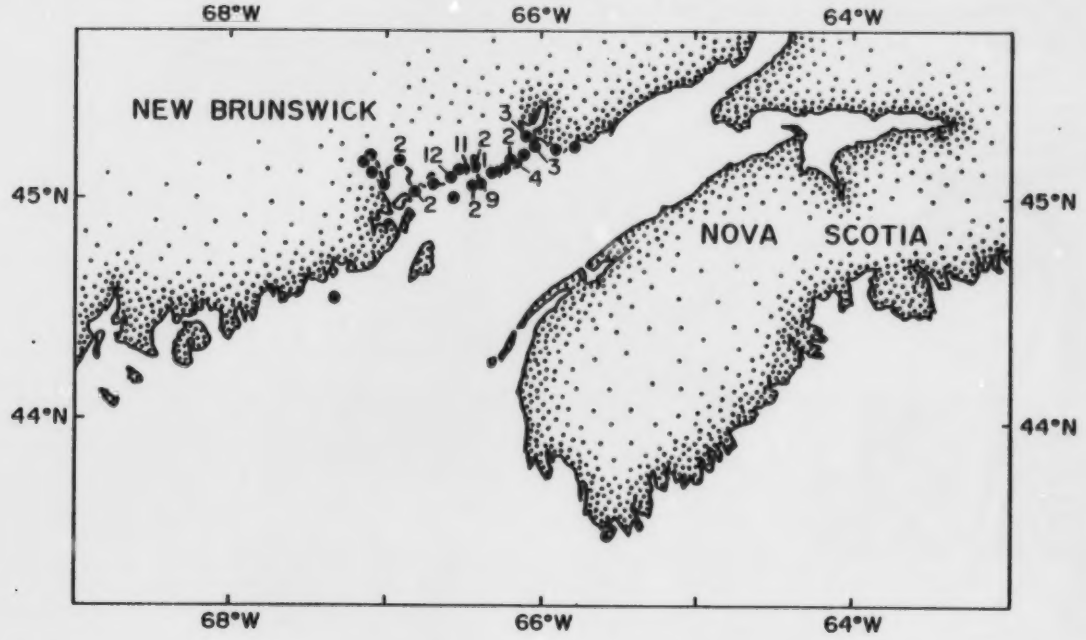


Figure 11. Recovery positions for bottom drifter returns. Numbers indicate multiple returns.

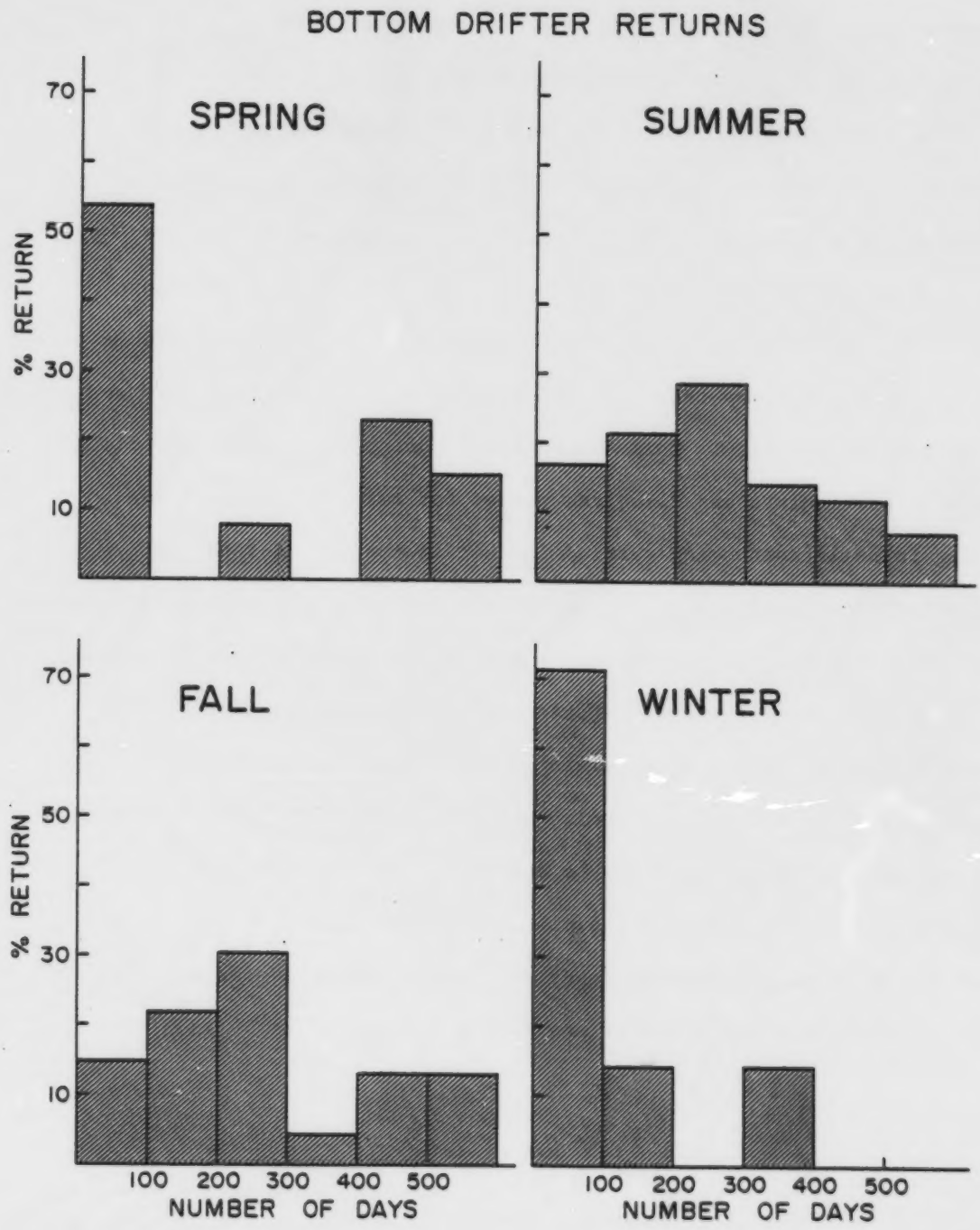


Figure 12. Bottom drifter return percentages as a function of drift time, in days, for different seasons.

half in some cases, suggests a long residence time in the Maces Bay-Passamaquoddy Bay area prior to their recovery and indicates that this region may be one of pronounced sediment deposition and resuspension.

The ocean drifter results are generally consistent with a pronounced spring freshwater outflow along the northwestern side of the Bay of Fundy attaining a maximum in the late spring or early summer. In the winter, eddies within the Bay of Fundy are closed and there is a general northwest to the southwest drift. The proximity of the freshwater plume to the Point Lepreau NGS may result in the release of radionuclides into seawater of reduced salinity. This could affect the speciation and partitioning between particulate and dissolved phases for released radionuclides, resulting in pollutants being carried further in the Bay of Fundy before mixing with higher salinity bottom water. In general there will be greater mixing and more efficient dispersion of pollutants entering less stratified water in fall and winter months. During periods of increased stratification in the spring, the vertical mixing of conservative radionuclides (such as tritium) will be reduced but their horizontal transport and impingement on the coastal intertidal zone may be enhanced. Bottom drifter results indicate that particle-reactive radionuclides (such as Pu, Cs) may be deposited in the Maces Bay-Passamaquoddy Bay region in the summer and fall and may be redistributed by winter and spring storms.

3.0 CHEMICAL OCEANOGRAPHY

Radionuclides discharged into the ocean from reactors and nuclear fuel reprocessing plants are generally point source releases into coastal waters. They may be in the form of primary or secondary cooling waters, treated low level wastes, or accidental discharges (cooling water spills, ruptured fuel elements). The physico-chemical forms of the artificial radionuclides entering the oceans from these various sources are diverse and generalizations concerning their behavior and fate in the marine environment are not applicable to all marine regimes. For example, the physico-chemical forms of many radionuclides discharged in coastal waters that contain high concentrations of suspended sediments, plankton and dissolved and particulate organic carbon would be grossly different than the same radionuclides deposited into the open ocean. Radionuclides discharged from reactors often enter the oceans in different chemical forms from those of their stable analogues present in the receiving waters. If equilibration is slow between the different chemical forms, the behavior of the radionuclides and their stable analogues in the marine biosphere may exhibit significant differences.

The purpose of the chemical oceanographic segment of this program was to characterize the general water column chemistry of the western Bay of Fundy and establish baseline levels of radionuclides and their stable analogues in order to predict the transport properties of radionuclides released from the Point Lepreau NGS. Important hydrographic parameters (temperature, salinity), nutrients, trace metals and suspended particulate matter (SPM) levels were examined during the pre-operational phase of this program. In addition, measurements were made to establish baseline levels for Cs-137, Sr-90, H-3, Pb-210, plutonium and gamma-emitting radionuclides

both in the water column and sediments. These measurements provide an important pre-operational baseline for the assessment of post-operational releases of radioactivity, in addition to radioactive tracer information on water circulation patterns, dispersion rates and particle transport processes in the Point Lepreau region.

3.1 Radionuclides - Water Column

A series of oceanographic cruises were conducted in the western Bay of Fundy to characterize chemical oceanographic parameters. The seawater sampling procedures and the analytical methods used at the AERU laboratory facilities have been outlined in previous reports in this series. In brief, fifty liter seawater samples were collected at surface and bottom water depths and the seawater samples were passed through (1) a 0.3 micron filter, (2) a Chelex-100 ion exchange resin column and (3) a KCFC (potassium cobalt ferrocyanide) column (Fig. 13). The filters were used to determine SPM levels and subsequently analyzed for gamma-emitting radionuclides. Cations, including activation products such as Co-60 and Mn-54, were eluted from the Chelex-100 column with weak acid. The eluate was then analysed for gamma-emitting radionuclides. The KCFC column material was analysed for Cs-137 and twenty liters of filtered seawater was saved for later Sr-90 analysis. Samples were also collected and analysed for nutrients, salinity, trace metals and tritium. The sampling matricies and nuclear instrumentation used in this program are summarized in Table 1.

3.1.1 Cs-137 and Sr-90 Activities in Seawater

Among the most environmentally hazardous radionuclides entering marine systems are those with relatively long half-lives which are rapidly

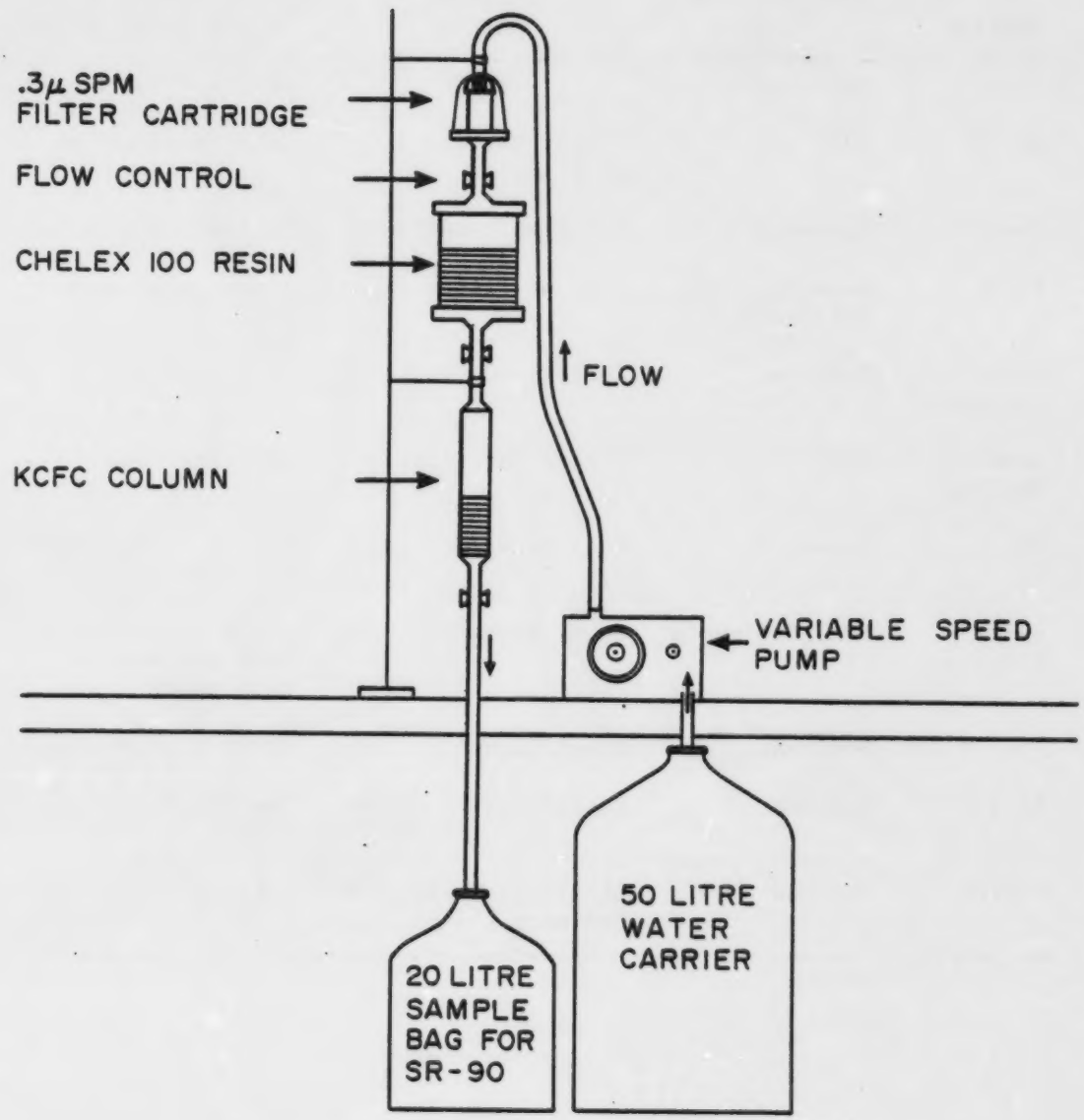


Figure 13. Schematic diagram of column used to concentrate radionuclides from seawater.

Table 1. Summary of nuclear instrumentation used in the AERU Laboratory.

NUCLIDE	ENVIRONMENTAL PHASE	COUNTING MATRIX	DETECTION SYSTEM
Cs-137	Seawater	KCFC	GeLi and NaI Detectors
Cs-137	SPM	Filter Cartridge (0.3 μm)	GeLi Detector
Cs-137	Sediment	Dried Bulk Sediment	GeLi and NaI Detectors
Sr-90	Seawater, SPM and Sediment	Y-90 on Filter Paper	Alpha/Beta Counter
Gamma Emitters	Seawater	Chelex-100 Eluate	GeLi Detector
Gamma Emitters	SPM	Filter Cartridge (0.3 μm)	GeLi Detector
H-3	Seawater	Gel (Water/Cocktail)	Liquid Scintillation Counter
Pb-210 (Po-210)	SPM, Sediments	Ag,Ni Discs	Alpha Spectrometer (Surface Barrier Detectors)
Pu-239,240	Sediment	Steel Disc	Alpha Spectrometer
Ra-226	Sediment	Ra-222 Gas	Gas Phase Scintillation Counter
Ra-226	Sediment	Bi-214 in Dried Bulk Sediment	GeLi Detector

and efficiently incorporated into the marine biosphere. Although most of the major fission products in nuclear reactor fuel waters do not fall in this category, Sr-90 ($t_{1/2}$ = 28.8y) and Cs-137 ($t_{1/2}$ = 30y) have long half lives and long residence times in the ocean and both are biologically important. Sr-90 follows the same metabolic pathways as calcium which provides a relatively direct pathway to man, while Cs-137 has high radiotoxicity and has proved to be useful as a tracer for global scale processes. In addition, they are the only fission products which will be released from the Point Lepreau NGS that are now generally present at detectable levels in the environment as a result of nuclear weapons fallout. The pre-operational program has attempted to establish the background levels of Cs-137 and Sr-90 in the Bay of Fundy in order to recognize future reactor-derived inputs and the presence of these radionuclides in the ecosystem allows us to study the transport processes affecting these radionuclides and their net accumulation in different environmental reservoirs.

The Cs-137 results for seawater samples collected during the April 1981 cruise (#81-010) are listed in Tables 2 and 3. Radioactivity measurements are reported in units of Becquerals (Bq). Conversion factors for commonly used radioactivity units are listed in Appendix 1. The mean Cs-137 activity in seawater for all stations excluding the anchor station was 3.50 mBq/l with a range of 2.67 mBq/l to 4.15 mBq/l. This mean Cs-137 activity is comparable to means from previous cruises in this series; 4.0 mBq/l in May, 1979, 4.24 mBq/l for June, 1980 and 2.7 mBq/l for the October, 1979 (Table 4) (Bishop et al., 1980, Smith et al., 1981, 1982). Surface ocean Cs-137 values have been reported in the range of 1 to 20 mBq/l and decrease to background levels with increasing water depth below

Table 2. Cs-137 and tritium results for water samples collected on Cruise 81-010. soluble particulate

Sample No.	Station	Depth (m)	Salinity °/₀₀	SPM mg/l	Cs-137(mBq/1) Soluble	Cs-137(mBq/1) particulate	H-3 Bq/1
81-07901	1	150	33.041				
07902	1	75	32.614				
07903	1	5	32.350				
07904	2	175	32.982				
07905	2	99	32.713				
07906	2	2	32.275				
07907	3	160	32.846	0.80	3.33 ± 0.52		1.77 ± 1.9
07908	3	160	32.846	0.60	3.80 ± 0.50		
07909	3	84	32.750	0.81	3.57 ± 0.57		1.61 ± 1.9
07910	4	19	31.833	14.71	3.40 ± 0.52		
07911	4	1	31.835	3.21			
07912	5	39	31.835				
07913	5	1	26.468	5.43	2.67 ± 0.70	<1.6	
07914	6	70	32.186	5.56	3.17 ± 0.55		
07917	6	1	29.507	10.91	3.40 ± 0.50		<1.5
07918	7	54	31.917				
07919	7	24	31.864				
07920	7	2	28.885				
07921	8	38	31.853				
07922	8	19	31.821				
07923	8	2	27.681				
07924	9	26	31.815	9.54			
07925	9	1	31.118	22.24			
07926	10	66	31.978				
07927	10	34	31.951				
07929	11	68	32.094				
07930	11	36	32.078	12.23			
07931	11	2	29.933	10.04			
07932	12	80	31.990				
07933	12	1	31.815				
07934	13	55	31.931				
07935	13	28	31.862				
07936	13	1	31.406				
07937	14	39	31.931				
07938	14	24	31.862				
07939	14	1	31.867				
07941	15	68	31.117				
07942	15	34	32.758				
07943	15	1	31.716				
07944	16	37	31.936	9.56	3.28 ± 0.57		
07945	16	1	31.870	2.37	3.13 ± 0.50		
07946	17	39	31.891				
07947	17	19	31.816				
07948	17	1	31.662				

Table 2. Cont'd.

Sample No.	Station	Depth (m)	Salinity °/₀₀	SPM mg/l	Cs-137(mBq/1) Soluble	Cs-137(mBq/1) particulate	H-3 Bq/1
81-07949	18	17	31.885	1.66			
07950	18	1	31.803	1.54	3.28 ± 0.50		
07951	19	29	32.038	2.67	3.17 ± 0.52		
07952	19	2	31.616	1.86	4.15 ± 0.53		
07953	20	45	32.064				
07954	20	24	31.961				
07955	20	2	31.621				
07957	21	40	32.174	1.95	3.87 ± 0.52	<1.2	2.73 ± 1.9
07958	21	2	31.727	1.50	3.78 ± 0.52		3.02 ± 1.9
07959	22	57	31.712				
07960	22	24	31.601				
07961	22	2	32.215				
07963	23	66	32.215	10.62	3.45 ± 0.50		
07964	23	1	31.744	1.24	3.32 ± 0.37	<1.2	
07965	24	59	32.149	15.99	3.60 ± 0.50		1.89 ± 1.9
07966	24	1	31.684	1.17	3.27 ± 0.50		2.49 ± 1.9
07967	25	88	32.220				
07968	25	44	32.139				
07969	25	1	31.161				
07970	26	98	32.533				
07971	26	54	32.367				
07972	26	1	30.941				
07973	27	109	32.608				
07974	27	58	32.447				
07975	27	1	31.286				
07976	28	124	32.687				
07977	28	59	32.453				
07978	28	2	31.049				
07979	29	122	32.762				
07980	29	59	32.489				
07981	29	2	31.237				
07983	30	128	32.644	2.64	3.45 ± 0.48		
07984	30	2	31.089	1.09	3.67 ± 0.52		
07985	31	139	32.650				
07986	31	70	32.603				
07987	31	2	31.383				
07988	32	152	32.760				
07989	32	80	32.666				
07990	32	2	31.304				
07991	33	113	32.630	3.17	3.93 ± 0.47		
07992	33	1	31.636	0.85	3.98 ± 0.57		
07993	34	93	32.337	9.58	3.17 ± 0.52	<1.1	
07994	34	40	31.497	1.10	3.42 ± 0.52	<1.1	
07998	37	40	32.080	8.14	3.68 ± 0.50	<1.1	
07999	37	2	31.598	9.07	3.45 ± 0.50		

Table 2. Cont'd.

Sample No.	Station	Depth (m)	Salinity °/..	SPM mg/l	Cs-137(mBq/l) Soluble	Cs-137(mBq/l) particulate	H-3 Bq/l
81-08000	38	49	32.092				
08001	38	25	31.751				
08002	38	2	31.593				
08003	39	61	32.152	9.49	3.65 ± 0.52	<1.4	
08004	39	2	31.550	8.78	3.78 ± 0.52		
08007	40	92	32.368				
08008	40	55	32.018				
08009	40	1	31.597				
08010	41	118	32.086				
08011	41	63	31.958				
08012	41	1	31.492				
08013	42	90	32.222	1.50	3.85 ± 0.38	<1.4	
08014	42	1	31.744	0.97	3.88 ± 0.48		
08015	43	100	31.982				
08016	43	54	31.979				
08017	43	1	31.977				
08018	44	132	32.524	1.75	3.27 ± 0.50	<1.1	
08019	44	1	31.502	1.20	3.63 ± 0.50	<1.1	
08020	45	124	32.862				
08021	45	65	32.339				
08022	45	1	31.334				
08023	46	96	32.349				
08024	46	75	32.350				
08025	46	48	32.323				
08026	46	15	31.324				
08027	46	2	29.808				
08028	47	85	32.245				
08029	47	51	32.236				
08030	47	22	31.856				
08031	47	13	30.741				
08032	47	4	29.721				
08033	47	2	26.072				

Other radionuclides: Pu 239/240

Sample No.	Station	Depth (m)	SPM (mg/l)	Total (mBq/ℓ)	Particulate (mBq/ℓ)	Pu(+4) (mBq/ℓ)	Pu(+5) (mBq/ℓ)
81-7962	23	66	2.03	0.033 ± .001	0.013 ± .001	0.006 ± .001	0.014 ± .001
81-7982	30	125	0.80	0.023 ± .001	0.004 ± .001	0.004 ± .001	0.014 ± .001
81-8063	62	37	18.44	0.070 ± .004	0.052 ± .004	0.004 ± .001	0.016 ± .001

Table 3. Surface and bottom Cs-137 and H-3 activity levels measured in surface and bottom water samples collected at anchor station (#50-78) during cruise 81-010.

Sample	Cast Time	Water Depth(m)	Salinity (°/₀₀)	SPM (mg/1)	Cs-137(mBq/1) soluble	Cs-137(mBq/1) particulate	H-3 (Bq/1)
81-8037	7:30	35	31.756	15.34	3.03 ± 0.53		3.4 ± 1.8
8038		1	30.065				2.4 ± 1.8
8039	8:30	37	31.819				2.3 ± 1.8
8040		1	30.450	16.54	3.78 ± 0.38		
8041	9:30	39	31.771	28.50	3.68 ± 0.53		
8042		1	30.923				
8043	10:30	39	31.772				
8044		1	30.957	9.29	3.60 ± 0.50	<1.0	2.44 ± 1.8
8045	11:30	40	31.708	13.11	3.15 ± 0.50		
8046		1	30.892				
8047	12:30	41	31.803				
8048		2	31.421	8.92	3.48 ± 0.50	<1.4	
8049	13:30	41	31.902	3.90	3.17 ± 0.53		
8050		2	31.420				
8051	14:30	40	31.932				
8052		2	31.440	4.14	3.58 ± 0.52		
8053	15:00	39	31.896	7.60	4.13 ± 0.50	<0.9	
8054		2	31.535				
8055	16:30	37	31.702				
8056		2	31.293	3.03	4.21 ± 0.53	<1.2	
8057	17:30	37	31.711	7.27	4.20 ± 0.53	<1.2	
8058		2	31.601				
8059	18:30	38	31.640				
8060		1	30.177	1.68	3.42 ± 0.50	<1.5	
8061	19:30	37	31.649		3.07 ± 0.50		
8062		1	29.931				
8064	20:00	38	31.749				
8065		1	29.095	3.09	3.70 ± 0.53	<1.5	
8066	21:30	35	31.891		3.38 ± 0.45		
8067		1	31.364				
8068	22:30	37	31.686				
8069		1	30.100	1.89	3.87 ± 0.51	<1.3	
8070	23:30	39	31.868		3.82 ± 0.51		
8071		1	31.383				
8072	0:30	37	31.660				
8073		2	30.123	2.13	3.90 ± 0.53	<1.0	
8074	1:30	37	31.900		3.20 ± 0.52		
8075		2	30.900				
8076	2:30	37	31.821				
8077		2	29.428	1.47	3.92 ± 0.50	<1.2	
8078	3:30	38	31.804		3.63 ± 0.50		
8079		2	30.925				
8080	4:30	36	31.208				
8081		2	30.121	1.95	4.02 ± 0.52	<1.1	

Table 3. Cont'd.

Sample	Cast Time	Water Depth(m)	Salinity (°/₀₀)	SPM (mg/1)	Cs-137(mBq/1) soluble	Cs-137(mBq/1) particulate	H-3 (Bq/1)
81-8082	5:30	37	31.693		3.87 ± 0.50		
8083		2	30.296				
8084	6:30	34	31.563				
8085		3	28.085		3.88 ± 0.45		
8086	7:30	33	31.586		3.10 ± 0.52		
8087		3	29.466				
8088	8:30	34	31.416				
8089		3	29.857		4.02 ± 0.55		
8090	9:30	34	31.648		3.03 ± 0.50		
8091		3	29.282				
8092	10:30	38	31.527				
8093		3	28.521		3.17 ± 0.37		
8094	11:30	37	31.572		4.12 ± 0.48		
8095		3	27.975				

the mixed surface layer (Bowen et al., 1974, Murray et al., 1978). As has been observed in previous cruises in the Bay of Fundy, there is no strong correlation between Cs-137 activity and either water depth or geographical location. These observations are not unusual and are indicative of the efficient tidal mixing in the Bay of Fundy. During the two 1979 cruises a weak ($r = -0.55$, $n = 80$) inverse correlation, between salinity and Cs-137 activity, was found. In contrast, the June 1980 cruise (80-018) displayed no significant correlation between salinity and Cs-137 activities (Smith et al., 1981) and there was no significant correlation between salinity and Cs-137 activities for cruise 81-010 ($r = -.23$, $n = 59$). In both the 1980 and 1981 cruises the Cs-137 activities and salinities were within narrow ranges and the possible effects of freshwater inputs were not resolved in either case. Suspended particulate matter (SPM) concentrations were slightly higher in bottom water (averaging 6.44 mg/l) than in surface water (averaging 4.70 mg/l), probably as a result of sediment resuspension phenomena.

Surface and bottom water samples were collected every hour for a 28 hour period at a designated anchor station (stations 50-78, each cast was considered a separate station), located several hundred metres off the reactor outfall at Point Lepreau. Surface and bottom water values for Cs-137, salinity, SPM and tidal cycle are illustrated in Figs. 14 and 15. Salinities for bottom water samples are within a narrow range of values and exhibit minimal tidal dependence. Surface water salinities are more variable and display a strong tidal influence with a noticeable drop in salinity at low tide. The position of the anchor station places it within the influence of the freshwater plume originating from the Saint John River and as illustrated in Figs. 6 and 7, salinities in the freshwater plume are

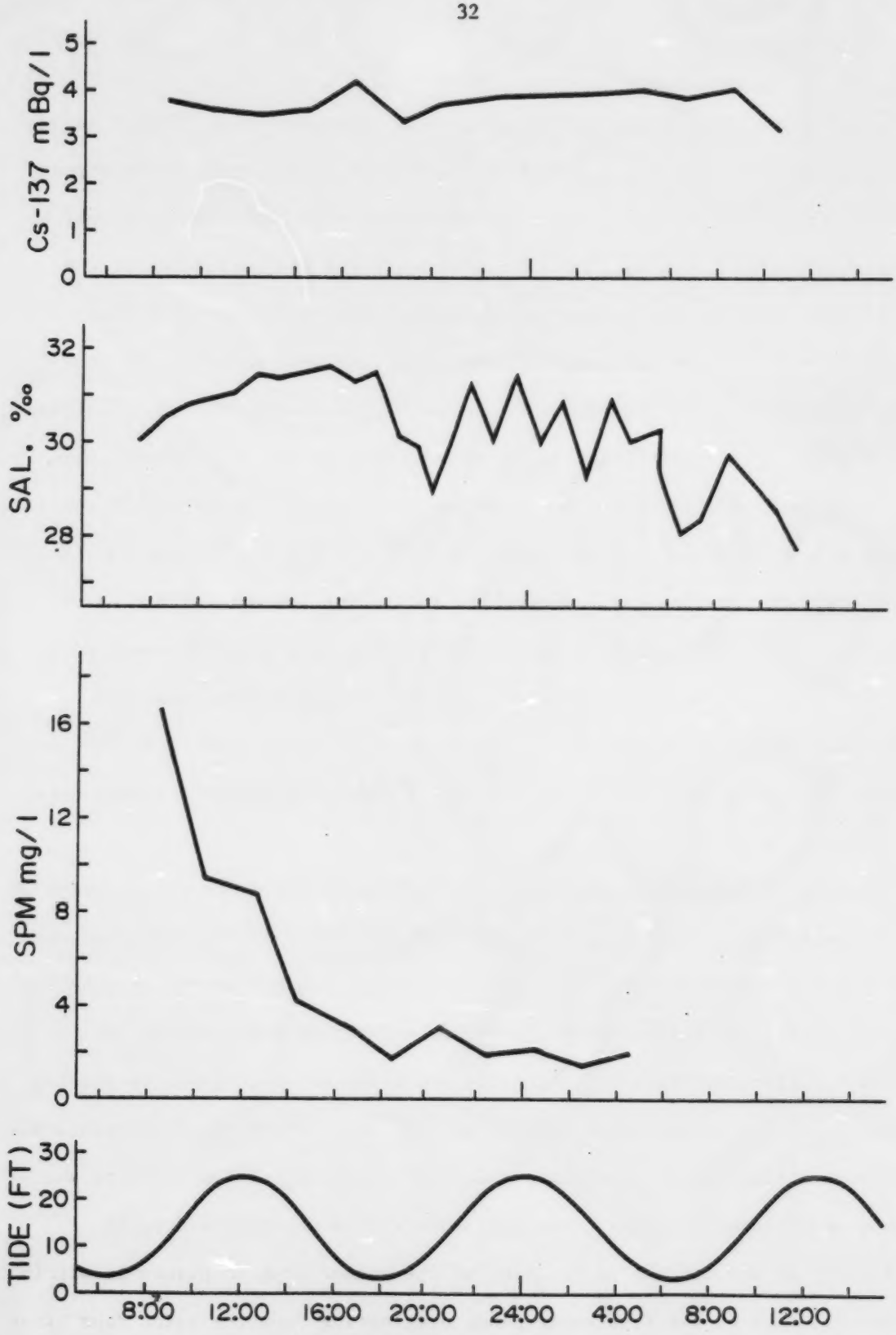


Figure 14. Cesium -137, SPM and salinity plotted as a function of time for surface water samples collected at anchor station 50 during cruise 81-010.

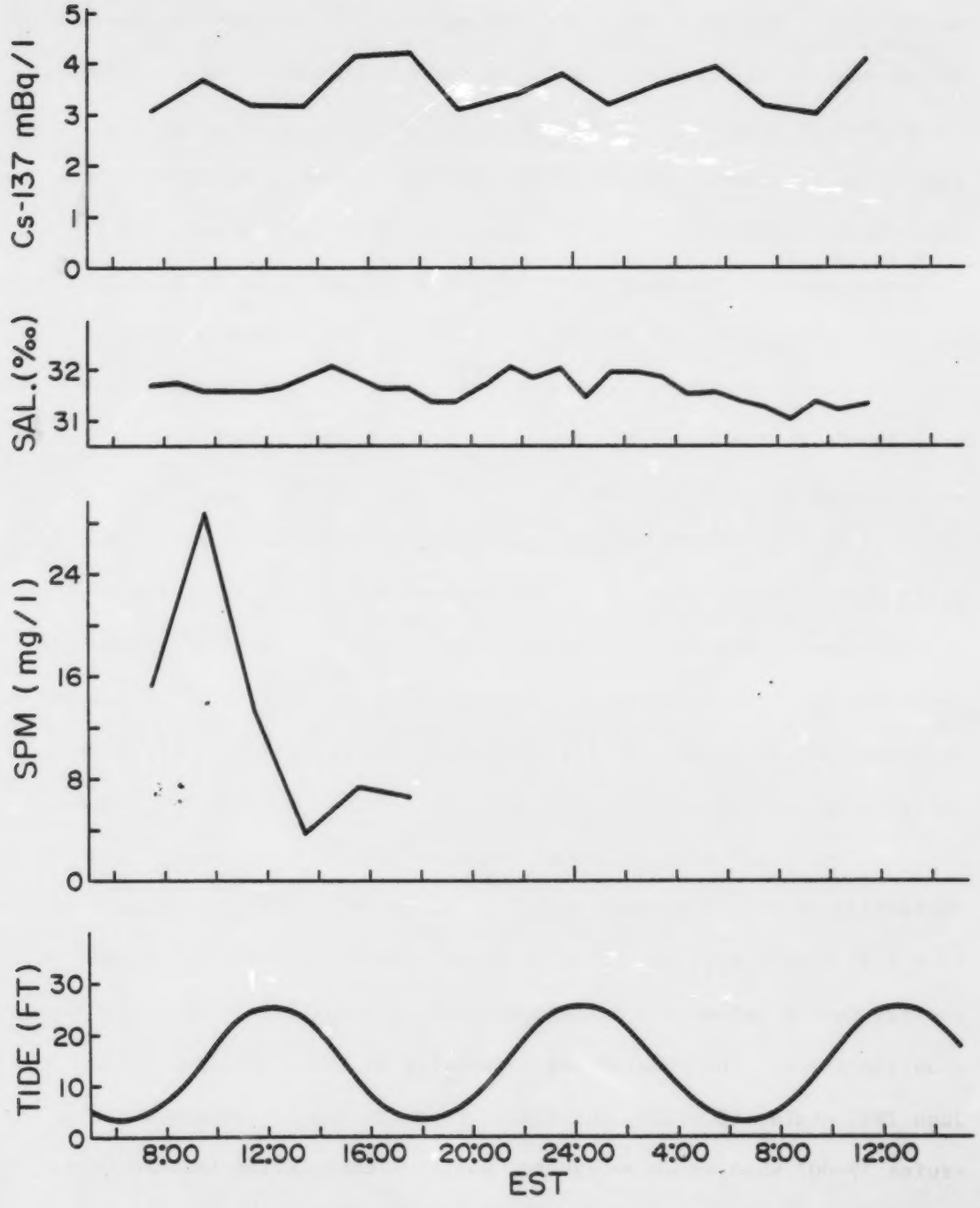
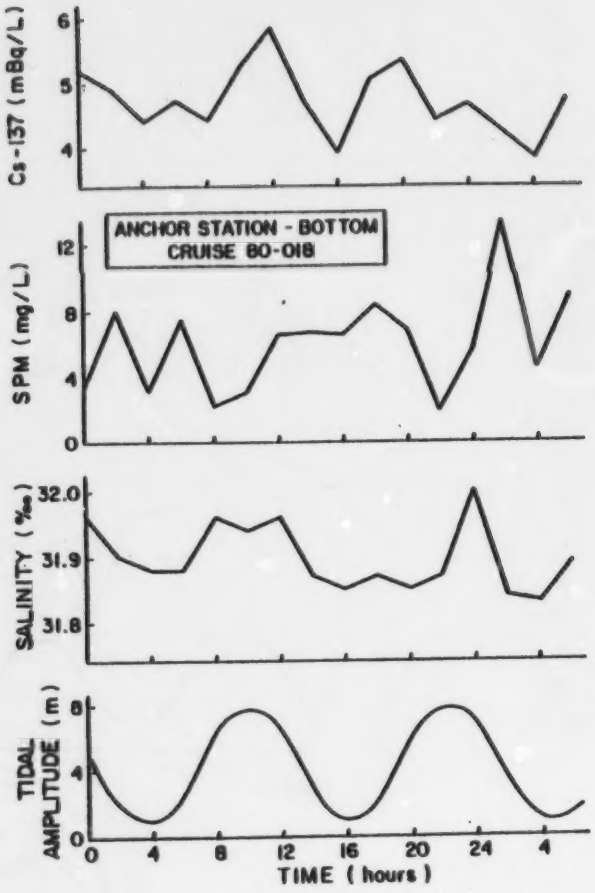


Figure 15. Cesium -137, SPM and salinity plotted as a function of time for bottom water samples collected at anchor station 50 during curise 81-010.

less than 29 °/₀₀. Suspended particulate matter values at the anchor station are lower in surface water samples (average = 4.92 mg/1) than in bottom water samples (average = 12.62 mg/1) and higher levels measured in bottom samples are probably due to sediment resuspension and transport in the nepheloid layer. The SPM results for the previous cruises exhibit a wide range of values. Suspended particulate values up to 28 mg/1 have been observed in bottom water samples with minimum SPM levels measured in the 2 to 4 mg/1 range. The sharp peaks in the SPM data can be related to changing tidal conditions and current velocities. The large SPM spike (28 mg/1) observed for bottom water sample at the anchor station during cruise 81-010 corresponds to the incoming tide (Fig. 15) and an eastward flowing current of approximately 1 m/sec (Bugden, 1983). Smaller SPM values, ranging from 12 to 16 mg/1 were observed for the June cruise 80-018 (Fig. 16) and the April cruise 79-007 (Fig. 17). The smaller SPM peaks for cruise 80-018 corresponded to eastward moving currents of 0.2 to 0.4 m/sec. The data indicates that resuspension of fine-grained material from the Maces Bay - Passamaquoddy Bay area with transportation in eastward moving bottom currents as the tide rises may be responsible for observed SPM peaks.

The Cs-137 data collected during cruise 81-010 displays similar variability for both surface ($\bar{x}$ = 3.64 mBq/1, std. dev. = 0.34) and bottom ($\bar{x}$ = 3.51 mBq/1, std. dev. = 0.34) water samples with the difference in average Cs-137 values for surface and bottom being less than the 2 sigma counting error. The results are comparable to those obtained during the June 1980 cruise (80-018), but differ from the results obtained during cruise 79-007 when an out-of-phase, positive correlation between Cs-137 activity and SPM level was noted. The data collected during the spring cruise was heavily influenced by the extremely high river discharge and



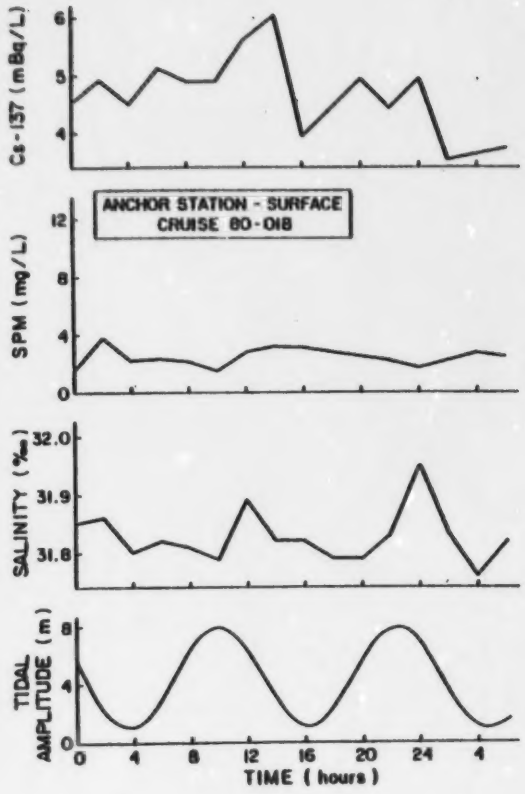


Figure 16. Cesium -137, SPM and salinity as a function of time for surface and bottom water samples collected at the anchor station during cruise 80-018.

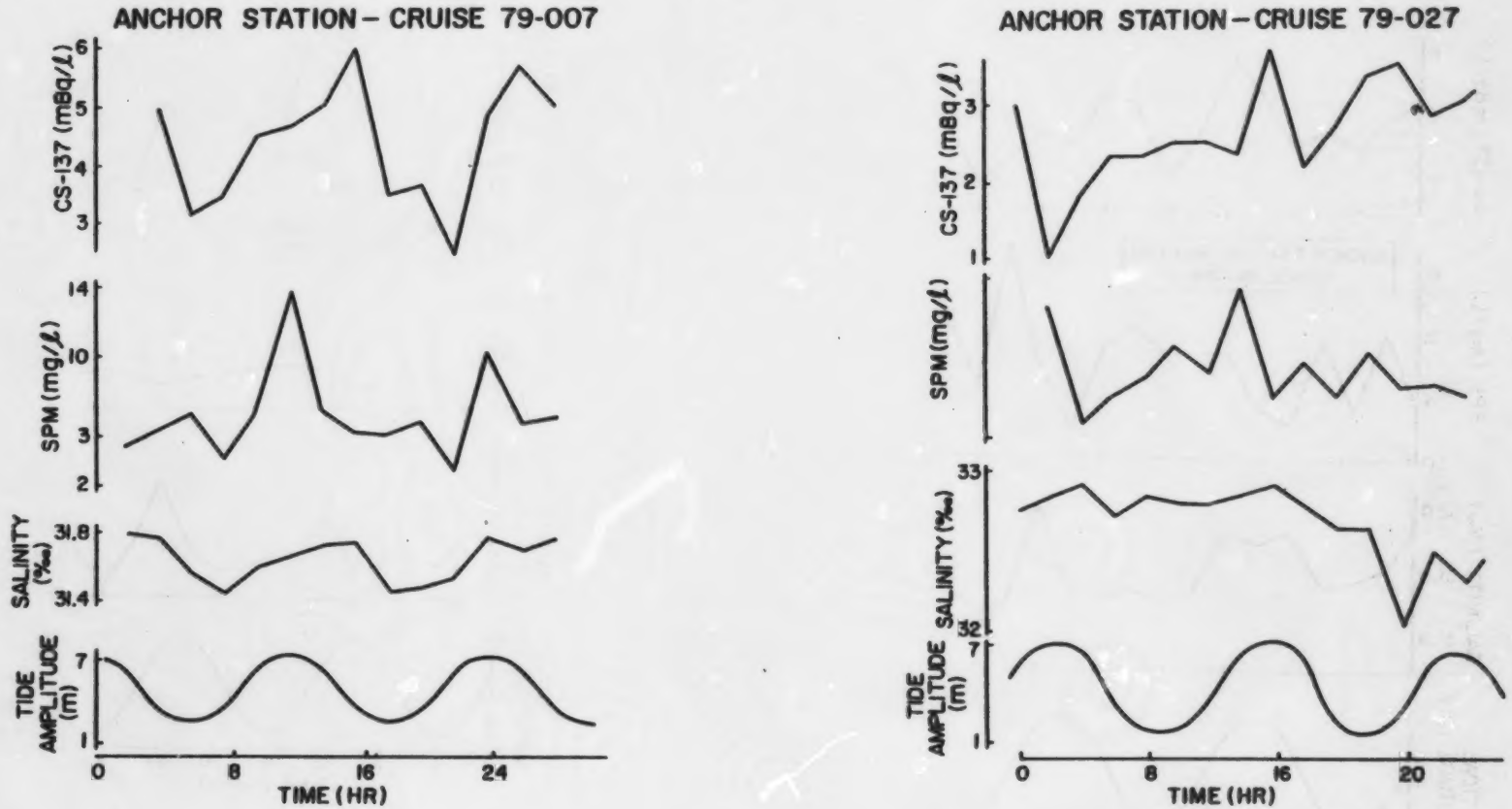


Figure 17. Cesium-137, SPM and salinity as a function of time for surface and bottom water samples collected at anchor station, for cruises 79-007 and 79-027 respectively.

water column stratification observed that year at the anchor station. It is possible that this phenomena may be related to enhanced inputs of particulate Cs-137 originating from resuspended sediments or river borne particulate material.

The range in Cs-137 activities at the anchor station (range = 3.03 mBq/l to 4.21 mBq/l) is similar to that found for the entire sampling area (range 2.67 mBq/l to 4.21 mBq/l) and is evidence to support the contention that tidal mixing will produce rapid dilution of any radioactive plume released from the reactor cooling water outfall.

Sr-90 activities measured on samples collected during cruise 79-007 ranged from 1.3 mBq/l to 4.8 mBq/l, with a mean value of 3.0 mBq/l (Table 4). These values were slightly lower than typical coastal water values for Sr-90 and exhibit no significant correlation with salinity, water depth, geographical location or SPM concentrations. The mean Cs-137/Sr-90 activity ratio of 1.42 (S.D. = 0.53) is close to the fallout ratio of 1.5 and is more typical of open ocean ratios than the mean coastal ratio of 1.07 measured by Bowen et al. (1974). The small quantity of experimental data and the large analytical uncertainties in the Sr-90 measurements probably contribute to the observed deviation from the expected value.

3.1.2 Gamma-Emitting Radionuclides

Neutron activation products of transition metals such as Zn-65, Co-60 and Mn-54 are environmentally important radionuclides resulting from both nuclear weapons fallout and effluents from reactor facilities. These radionuclides are concentrated to a high degree by the biosphere because they are physiologically esssential to marine organisms. Concentration factors from seawater on the order of 10^3 to 10^5 have been observed

Table 4 Average radionuclide values measured in the water column for all cruises.

		Radionuclide Concentrations (mBq/l)			
Cruise #	Date	Cs-137 part.	Cs-137 Dissolved	Sr-90	H-3
79-007	May 01-06, 1979	-	3.78* ± 0.68** n = 28	3.121 ± 1.12 n = 13	2700 ± 3000
79-027	Oct 24-28, 1979	-	2.60 ± 0.77 n = 24	-	1700 ± 2700
80-018	June 9-14, 1980	<0.81	4.25 ± 0.30 n = 43	-	<2300
81-500	Feb 23-26, 1981	<1.5	4.36 ± 0.53 n = 20	-	2840 ± 1200
81-010	Apr 14-21, 1981	<1.2	3.54 ± 0.31 n = 33	-	2210 ± 580

* Mean radionuclide values calculated using a single mean from the anchor station data.

** Error reported is the statistical error for mean value.

+ K_D for Cs-137 in particulate and dissolved phases for cruise 81-010 is $<1.3 \times 10^5$.

(Robertson, 1971).

The gamma-emitting radionuclides listed in Table 5 are concentrated from sea water by adsorption onto Chelex-100 (with the exception of I-131 and Cs-137). The average retention efficiency of 90% was determined using an NBS mixed gamma radionuclide standard. The detection limits in Table 5 are based on an assumed water volume of 36 l with the eluted samples counted for 1500 minutes, one month after the initial sample collection.

Gamma-emitting radionuclides were measured on SPM by direct counting of filter cartridges on the Ge(Li) detector. Standards were prepared by direct spiking of the filters with a mixed gamma radionuclide standard. The detection limits for each radionuclide listed in Table 5 were calculated assuming an SPM concentration typical of the Bay of Fundy of 4.7 mg/l, a seawater volume of 36 l and a counting time of 1500 minutes, one month after sample collection.

3.1.3 Tritium in Seawater

The major tritium inputs into the environment have been associated with fallout from nuclear weapons tests, in addition to a natural input pathway originating from cosmogenic interactions in the upper atmosphere. Environmental releases of tritium from the Point Lepreau NGS will be substantially greater than those for any other radionuclide under normal operating conditions and elevated levels of tritium associated with reactor releases should be detectable in a range of environmental phases. It is therefore important to assess the present, background levels of tritium in a variety of different environmental reservoirs.

Water samples collected for tritium analysis in the Bay of Fundy were distilled under vacuum and counted for 500 minutes using a liquid

Table 5. Detection limits for gamma emitting radionuclides in dissolved and particulate phases. Calculation based on seawater volume of 50 l, SPM concentration of 2.1 mg/l, and counting time of 1500 min one month after sample collection.

Nuclide	Half-life (Days, Years)	Dectection Limit Suspended Matter (mBq/l)	Dissolved (mBq/l)
Ag-110 m	250.4d	0.60	5.1
Ce-141	33.0d	1.70	11.6
Ce-144	284.4d	0.51	31.6
Co-58	71.2d	0.70	9.6
Co-60	5.27y	0.60	4.3
Cr-51	27.7d	10.00	64.5
Cs-134	2.06y	0.56	1.0
Cs-137	30.0y	0.60	0.5
I-131	8.06d	14.00	-
Mn-54	321.6d	0.60	3.7
Nb-95	35.2d	1.00	6.8
Ru-103	39.4d	1.00	6.6
Ru-106	368.0d	6.30	38.5
Zn-65	244.1d	1.20	29.6
Zr-95	65.0d	1.30	19.9

scintillation counter and methods outlined in previous reports. Baffin Bay deep water (<0.2 Bq/l), was distilled in the same manner and counted during each run as a tritium blank, counting efficiency standardization was performed for each sample set with a dilution of the National Bureau of Standards tritium standard reference material 4926-C. Counting efficiencies ranged from 25% to 26%. The detection limit (L_d) at the 95% confidence level was 1.5 Bq/l. The results for cruise 81-010 are given in Tables 2 and 3. The mean tritium activity for all samples except the anchor station was 1.87 Bq/l, and ranged from <1.5 Bq/l to 3.02 Bq/l. The average tritium value at the anchor station was 2.63 Bq/l and ranged from 2.3 Bq/l to 3.4 Bq/l with no difference between surface and bottom water samples. These values are comparable to those found on previous cruises, cruise 79-007, average of 2.7 ± 3.0 Bq/l and cruise 79-027; 1.7 ± 2.7 Bq/l (Table 4). The results for all cruises during the pre-operational program indicate no unusual seasonal or geographical trends for tritium results in the Bay of Fundy. It should be noted that present day environmental levels of tritium in the study area are at or below the detection limits for the analytical techniques employed by AERU.

3.1.4 Trace Metals and Nutrients

The oceans and particularly coastal waters are nonhomogeneous mixtures with regard to chemical composition. The combined mixing rates of substances introduced via rivers, circulation patterns which may favour the retention of dissolved chemicals in coastal waters, the relative abundances of particulate matter and the degree of biological activity are all instrumental factors in determining the chemical makeup of coastal marine waters. In the majority of coastal waters, precipitation and freshwater

runoff exceed evaporation from the ocean's surface. The less dense freshwater mixes in the surface layers and moves away from the coast, being replaced by a sub-surface input of more saline deep water. This process helps to maintain the chemical inhomogeneity of coastal waters by replacing chemicals lost from the surface layer by a variety of mechanisms (settling of particles, biological action), and the replenishment of depleted surface waters with available nutrients is vital to the high primary productivity characteristic of shelf seas.

It follows therefore that the degree of dispersion of effluent released from a reactor into coastal waters will depend on the physical and chemical characteristics of the receiving water mass and the chemical properties of the element itself. Therefore, studies on the distribution and behavior of stable elements not only aid in characterizing the receiving water mass but also provides useful information regarding the environmental pathways and fates of radioactive isotopes. Transport pathways for radionuclides will frequently be similar to those of their stable element analogues if they are present in the same chemical form. Nutrient measurements can be used to further characterize receiving waters and to provide valuable information on water circulation and bio-geochemical processes within the water column.

Water samples for nutrients, salinity, trace metals and SPM determinations were collected at several water depths using 12 l Niskin bottles modified for trace element collection (Bewers et al., 1974). Nutrient samples were drawn directly from the Niskin bottles and stored frozen for subsequent analysis by standard auto-analysis techniques. Unfiltered water samples were also collected for salinity determinations.

Samples for dissolved trace metals and SPM were treated as in

previous cruises (Smith et al., 1981). Trace metal analyses were carried out by Dr. P. Yeats and B. Amirault at the Bedford Institute using previously described solvent extraction and atomic absorption techniques (Bewers et al., 1976) with SPM concentrations being determined by gravimetric analysis of the filters (Sundby, 1974).

The results for nutrient analyses for cruise 81-010 are presented in Table 6 and lie within the ranges of values measured on previous Bay of Fundy cruises. Values for both silicate and salinities were within a much narrower range than previous cruises, particularly cruises 79-007, in April 1979 and 81-500 in February 1981 (Smith et al.). The inverse silicate-salinity relationship, present in previous cruises, was much weaker for cruise 81-010 ($r = -0.20$, $n = 102$) but values were within the range of previous data. The narrow ranges for both salinity and silicate values are thought to be the reason for this weak correlation.

Trace elements results for cruises 80-018, and 81-010 are presented in Tables 7 and 8. A summary of all trace element results for cruises 79-007, 79-027, 80-018, and 81-010 are presented in Table 9. Trace element concentrations measured during the pre-operational program are comparable to those measured in other coastal regions. Distributional patterns observed in previous cruises, ie. Fe, Mn, and Al having higher surface water concentrations and Ni, Co, Cu, Zn, and Cd exhibiting lower surface water concentrations were not as strongly pronounced during cruises 81-010 and 80-018. For both cruises, surface concentrations of Fe, Mn and Al were higher than bottom water concentrations, indicative of a continuation of the commonly observed, estuarine removal of Fe, Mn and Al to the sediments. For Ni, Co, Cu, Zn and Cd, bottom and surface concentrations fall in the same range, as illustrated in Table 9. Mean trace metal

Table 6. Nutrient results for cruise 81-010.

Sample No.	Station	Depth (m)	Temp (°C)	Silicate µg-at/l	Phosphate µg-at/l	Nitrate µg-at/l
81-07901	1	150	4.89	6.83	1.04	9.17
07902	1	75	4.51	8.16	1.26	10.57
07903	1	5	4.08	7.56	1.13	9.39
07904	2	175	4.82	7.11	1.21	9.58
07905	2	99	4.51	7.23	1.02	9.29
07906	2	5	4.00	8.55	1.18	10.07
07907	3	160	4.79	6.51	1.22	9.01
07909	3	84	4.33	9.07	1.40	11.78
07910	4	19	2.60	6.73	1.11	8.41
07911	4	1	2.43	8.66	0.99	8.45
07912	5	39	2.58	7.73	1.18	9.35
07913	5	1	2.52	9.48	0.81	6.94
07914	6	70	3.59	7.81	1.14	9.48
07917	6	1	2.36	12.49	1.16	11.46
07918	7	54	2.71	7.52	1.18	8.81
07919	7	24	2.61	7.32	1.08	8.87
07920	7	2	2.43	10.82	0.95	10.12
07921	8	38	2.63	8.82	1.35	10.60
07922	8	19	2.60	6.92	1.17	8.40
07923	8	2	2.52	14.29	1.09	11.25
07924	9	26	2.63	7.65	1.10	8.96
07925	9	1	2.52	8.52	1.25	9.31
07926	10	66	2.92	6.79	1.52	8.63
07927	10	34	2.83	7.34	1.52	8.95
07928	10	2	2.40	7.77	1.61	8.43
07929	11	68	3.23	6.76	1.19	8.36
07930	11	36	3.22	8.42	1.23	10.14
07931	11	2	2.39	11.50	1.11	10.98
07932	12	80	3.33	7.72	1.22	9.64
07933	12	1	2.70	11.44	1.16	11.04
07934	13	55	2.83	7.16	1.05	8.44
07935	13	28	2.64	7.22	1.11	8.43
07936	13	1	2.62	8.80	1.10	9.84
07937	14	39	2.71	7.76	1.26	9.06
07938	14	24	2.73	7.52	1.17	8.63
07939	14	1	2.73	8.41	1.25	9.74
07941	15	68	2.99	8.22	1.25	9.94
07942	15	34	2.84	5.49	1.19	6.63
07943	15	1	3.10	8.86	1.24	9.73
07944	16	37	2.90	8.69	1.34	9.95
07945	16	1	2.98	6.88	1.33	8.06
07946	17	39	3.02	6.57	1.18	7.57
07947	17	19	3.12	9.29	1.23	9.98
07948	17	1	3.52	7.20	1.28	7.70
07949	18	17	3.12	9.62	1.56	10.58
07950	18	1	3.39	7.84	1.17	8.47

Table 6. Cont'd.

Sample No.	Station	Depth (m)	Temp (°C)	Silicate µg-at/l	Phosphate µg-at/l	Nitrate µg-at/l
81-07951	19	29	3.31	8.62	1.27	9.69
07952	19	2	3.32	8.66	1.17	8.83
07953	20	45	3.31	9.29	1.19	10.35
07954	20	24	3.31	7.37	1.16	8.65
07955	20	2	3.69	7.29	1.02	7.39
07957	21	40	3.41	8.34	1.26	9.56
07958	21	2	3.72	7.08	1.04	7.38
07959	22	57	3.21	8.16	1.26	9.29
07960	22	24	3.29	9.29	1.19	9.75
07961	22	2	3.42	7.84	1.09	8.18
07963	23	66	3.21	5.82	1.05	6.86
07964	23	1	3.30	7.54	1.13	8.13
07965	24	59	3.13	6.76	1.17	7.89
07966	24	1	3.15	7.25	1.14	7.83
07967	25	88	3.31	7.75	1.19	9.10
07968	25	44	3.22	6.44	1.07	7.62
07969	25	1	2.83	8.38	2.06	8.85
07970	26	98	4.03	8.42	1.24	10.03
07971	26	54	4.19	9.14	1.46	11.10
07972	26	1	3.10	7.83	1.25	8.46
07973	27	109	4.12	6.78	1.06	7.86
07974	27	58	4.03	6.03	1.20	7.22
07975	27	1	3.45	8.67	1.24	9.24
07976	28	124	4.20	8.29	1.21	9.76
07977	28	59	4.02	5.23	0.98	6.18
07978	28	2	3.52	10.61	1.38	10.24
07979	29	122	4.33	8.47	1.22	10.16
07980	29	59	4.12	7.55	1.13	9.11
07981	29	2	3.41	8.41	1.07	8.78
07983	30	128	4.18	8.57	1.30	10.37
07984	30	2	3.77	9.04	1.12	9.45
07985	31	139	4.21	9.23	1.29	11.45
07986	31	70	4.13	9.00	1.22	10.74
07987	31	2	3.92	9.42	1.19	10.12
07988	32	152	4.39	8.69	1.20	10.42
07989	32	80	4.22	8.67	1.19	10.45
07990	32	2	3.53	10.48	1.22	11.03
07991	33	113	4.11	9.13	1.24	10.72
07992	33	1	3.12	9.09	1.15	9.49
07993	34	93	3.42	8.10	1.23	9.53
07994	34	40	3.42	9.26	1.23	9.82
07998	37	40	3.42	5.13	3.32	5.17
07999	37	2	3.53	4.91	3.28	4.78
08000	38	49	3.40	9.24	1.23	10.16
08001	38	25	3.45	7.01	1.08	7.44
08002	38	2	3.53	6.94	1.00	7.20

Table 6. Cont'd.

Sample No.	Station	Depth (m)	Temp (°C)	Silicate µg-at/l	Phosphate µg-at/l	Nitrate µg-at/l
81-08003	39	61	3.40	7.75	1.18	8.72
08004	39	2	3.53	8.75	1.24	9.13
08007	40	92	3.71	8.54	1.18	9.94
08008	40	55	3.43	6.73	1.10	7.60
08009	40	1	3.52	6.33	1.01	6.60
08010	41	118	3.51	8.54	1.28	9.68
08011	41	63	3.50	6.82	1.09	7.41
08012	41	1	3.73	6.93	1.06	7.09
08013	42	90	3.62	8.42	1.34	9.65
08014	42	1	3.30	7.7	1.15	8.32
08015	43	100	3.42	6.54	1.01	7.29
08016	43	54	3.43	6.55	1.08	7.24
08017	43	1	3.44	8.70	1.29	9.73
08018	44	132	4.30	8.41	1.28	9.88
08019	44	1	3.29	8.93	1.27	9.86
08020	45	124	4.61	8.74	1.34	10.83
08021	45	65	3.81	6.74	1.06	7.60
08022	45	1	3.13	8.26	1.06	8.51
08023	46	96	3.98	7.97	1.18	9.03
08024	46	75	3.98	7.47	1.14	8.60
08025	46	48	3.93	9.13	1.24	10.42
08026	46	15	3.22	9.07	1.15	9.35
08027	46	2	3.22	9.80	1.05	8.93
08028	47	85	3.86	9.05	1.35	10.28
08029	47	51	3.84	7.79	1.20	8.81
08030	47	22	3.90	9.19	1.22	10.04
08031	47	13	3.32	10.04	1.51	10.12
08032	47	4	3.21	9.93	1.18	9.29
08033	47	2	3.25	14.88	1.02	10.10
08037	50	35	2.82	7.99	1.11	8.40
08038	50	1	3.01	9.89	1.06	8.91
08039	51	37	2.84	11.32	1.28	10.68
08040	51	1	3.03	6.48	0.90	5.92
08041	52	39	2.89	9.09	1.21	9.64
08042	52	1	3.03	10.40	1.13	10.10
08043	53	39	2.89	8.27	1.21	8.84
08044	53	1	3.11	10.19	1.17	10.06
08046	54	1	3.10	10.38	1.53	10.62
08047	55	41	2.88	8.05	1.17	8.90
08048	55	2	3.10	7.52	1.05	7.85
08049	56	41	2.92	8.28	1.13	9.12
08050	56	2	3.09	5.34	0.90	5.80
08051	57	40	2.92	8.21	1.15	9.18
08052	57	2	3.17	8.48	1.14	8.93
08053	58	39	2.91	7.95	1.15	8.85
08054	58	2	3.02	7.71	1.05	8.15

Table 6. Cont'd.

Sample No.	Station	Depth (m)	Temp (°C)	Silicate µg-at/l	Phosphate µg-at/l	Nitrate µg-at/l
81-08045	54	40	2.88	8.74	1.18	9.28
08055	59	37	2.88	8.59	1.15	9.29
08056	59	2	3.22	8.50	1.11	8.89
08057	60	37	2.86	8.45	1.20	9.22
08058	60	2	3.04	6.53	1.01	7.09
08059	61	38	2.90	7.18	1.05	7.77
08060	61	1	3.35	11.11	1.14	10.29
08061	62	37	2.90	7.90	1.13	8.44
08062	62	1	3.41	7.79	0.94	7.12
08064	63	38	3.00	6.63	1.03	7.13
08065	63	1	3.49	10.18	1.00	8.51
08066	64	35	2.94	7.60	1.14	8.27
08067	64	1	3.05	10.26	1.19	10.32
08068	65	37	2.93	9.45	1.24	9.82
08069	65	1	3.47	9.07	1.07	8.37
08070	66	39	2.93	6.12	1.01	6.58
08071	66	1	3.02	8.79	1.14	8.98
08072	67	37	3.03	8.10	1.10	8.56
08073	67	2	3.50	11.55	1.14	10.38
08074	68	37	2.98	8.69	1.19	9.38
08075	68	2	3.38	7.75	1.06	7.67
08076	69	37	2.98	8.34	1.18	8.96
08077	69	2	3.42	11.76	1.18	10.25
08078	70	38	2.91	9.57	1.30	10.42
08079	70	2	3.32	10.82	1.18	10.47
08080	71	36	2.91	9.90	1.23	9.74
08081	71	2	3.08	8.86	1.06	8.17
08084	73	34	2.90	9.12	1.22	9.67
08085	73	3	3.40	11.08	0.99	8.43
08086	74	33	2.90	7.93	1.10	8.19
08087	74	3	3.47	12.76	1.19	11.26
08088	75	34	2.91	6.17	1.03	6.46
08089	75	3	3.23	7.90	0.97	7.07
08090	76	34	3.00	10.05	1.29	10.28
08091	77	3	3.42	8.91	1.02	7.57
08092	77	38	3.08	9.93	1.23	10.00
08093	77	3	3.62	10.70	1.00	8.51
08094	78	37	3.03	8.19	1.22	8.97
08095	78	3	3.75	12.45	1.00	9.39

Table 7. Trace element Results for cruise 80-018.

Sample No.	Station	Depth (m)	Fe (µg/l)	Mn (µg/l)	Co (µg/l)	Ni (µg/l)	Cu (µg/l)	Zn (µg/l)	Cd (µg/l)	Al (µg/l)
80-3548	49	43	1.62	1.05	0.009	0.35	0.76	1.40	0.040	1.10
3549	49	3	1.13	1.56	0.013	0.38	0.36	0.40	0.041	0.83
3487	26	2	1.35	2.36	-	0.40	0.39	-	0.049	0.58
3486	26	47	1.55	2.90	-	0.35	0.27	-	0.048	0.50
3449	14	113	3.23	0.81	-	0.33	0.15	1.09	0.023	2.59
3450	14	3.5	1.43	1.33	-	0.39	0.27	0.40	0.032	1.64
3509	30	35	1.55	2.14	-	0.39	0.17	0.27	0.025	0.72
3510	30	1	1.31	1.24	-	0.41	0.21	0.11	0.027	0.78
3553	49	3	1.68	1.50	-	0.33	0.29	1.12	0.019	1.10
3557	49	1	1.49	1.62	-	0.38	0.46	1.33	0.037	0.76
3555	49	2	1.54	1.35	-	0.35	0.23	0.91	0.020	0.46
3552	49	37	0.88	1.11	-	0.37	0.32	0.77	0.022	2.97
3551	49	2	1.01	1.98	-	0.30	0.37	0.55	0.044	1.71
3565	49	36	3.04	1.38	-	0.28	0.28	0.64	0.047	2,75
3577	49	2	1.18	1.11	-	0.27	0.23	0.17	0.031	1.58
3579	49	2	2.20	1.27	-	0.32	0.21	0.50	0.029	1.19
3578	49	40	2.11	1.39	-	0.43	0.23	0.50	0.035	0.59
3581	49	1	1.42	1.24	-	0.40	0.17	0.60	0.033	1.34
3582	49	37	1.32	1.67	-	0.43	0.17	0.36	0.032	1.04
3583	49	3	1.81	1.50	-	0.43	0.10	1.15	0.035	1.30
3440	12	95	0.62	1.41	-	0.31	0.25	0.47	0.029	-
3441	12	2	1.72	1.94	-	0.33	0.32	0.46	0.033	8.53
3472	23	69	0.84	0.70	-	0.31	0.23	0.54	0.028	0.72
2473	23	2	1.00	1.72	-	0.29	0.21	0.45	0.024	1.47
3519	33	44	1.92	1.45	-	0.31	0.26	0.22	0.012	0.76
3520	33	2	1.55	1.03	-	0.32	0.74	0.73	0.016	-
3548	49	43	1.75	1.05	-	0.31	0.66	0.95	0.019	1.10
3549	49	3	1.30	1.56	-	0.31	0.29	0.22	0.016	0.83
3420	7	3	1.67	1.64	0.018	0.40	0.34	1.04	0.014	2.53
3406	3	172	0.70	0.48	0.012	0.31	0.15	0.55	0.032	1.75
3407	3	3	3.22	1.38	0.018	0.32	0.21	1.47	0.035	3.54
3585	49	3	1.47	1.60	0.017	0.39	0.23	0.93	0.052	1.14
3426	8	100	0.65	0.90	0.007	0.30	0.14	2.87	-	1.74

Table 8. Trace element results for cruise 81-010.

Sample No.	Station	Depth (m)	Fe (µg/l)	Mn (µg/l)	Co (µg/l)	Ni (µg/l)	Cu (µg/l)	Zn (µg/l)	Cd (µg/l)	Al (µg/l)
81-7911	4	1	1.14	1.33	0.012	0.31	0.29	0.58	0.048	2.33
7913	5	1	4.12	3.61	0.018	0.36	0.22	-	0.029	5.47
7920	7	2	2.94	1.99	0.014	0.27	0.30	1.45	0.030	2.35
7925	9	1	1.40	0.70	0.008	0.28	0.28	0.45	0.031	2.14
7933	12	1	2.14	1.72	0.014	0.35	0.23	-	0.023	3.95
7958	21	2	1.23	0.74	0.008	0.25	0.29	0.64	0.036	3.18
7969	25	1	1.51	1.13	0.012	0.34	0.37	2.16	0.031	2.47
7992	33	1	1.45	1.03	0.013	0.31	0.27	0.30	0.027	1.98
7994	34	1	1.36	1.15	0.011	0.32	0.32	-	0.030	3.17
7999	37	2	2.16	1.48	0.023	0.32	0.36	-	0.029	2.52
8004	39	2	10.6	1.30	0.018	0.30	0.34	0.46	0.029	6.76
8012	41	1	1.91	0.97	0.016	0.34	0.35	2.32	0.038	2.28
8014	42	1	1.58	0.57	0.015	0.33	0.32	2.18	0.034	1.19
8027	46	2	2.77	1.86	0.020	0.37	0.59	0.72	0.068	3.25
8033	47	2	9.80	2.75	0.040	0.37	0.80	1.50	0.066	7.21
8037	50	35	1.47	0.50	0.011	0.31	0.30	0.37	0.026	1.30
8040	51	1	2.35	1.33	0.014	0.31	0.31	0.92	0.031	1.52
8045	54	40	1.02	0.72	0.011	0.31	0.28	4.69	0.036	2.56
8048	55	2	1.42	0.92	0.008	0.30	0.29	0.74	0.026	3.47
8053	58	39	0.98	0.67	0.009	0.35	0.31	2.44	0.028	1.73
8056	59	2	1.16	1.30	0.009	0.31	0.28	0.28	0.020	3.44
8061	62	37	1.00	1.11	0.014	0.32	0.33	0.52	0.029	2.28
8065	63	38	2.75	3.09	0.014	0.32	0.34	1.79	0.034	6.14
8070	66	39	1.63	0.73	0.010	0.31	0.30	1.12	0.034	1.37
8073	67	2	30.4	2.22	0.021	0.29	0.30	0.27	0.029	2.44
8078	70	38	1.08	0.51	0.012	0.33	0.20	-	0.026	1.41
8081	71	2	1.78	1.33	0.015	0.35	0.21	-	0.028	6.63

Table 9. Comparison of trace element results obtained for cruises 79-007, 79-027, 80-018, 81-010 ($\bar{X}$ = Mean; S = Standard Deviation; n = Number of Samples).

Cruise No.	Fe	Mn	Co	Ni	Cu	Zn	Cd	Ac	SPM
			µg/l						mg/l
79-007									
Surface									
$\bar{X}$	2.89	1.24	0.015	0.33	0.53	0.59	0.044	4.06	1.18
S	1.60	0.43	0.003	0.17	0.07	0.40	0.031	2.00	0.70
N	8	8	6	9	8	7	8	8	9
Bottom									
$\bar{X}$	1.67	0.69	0.014	0.60	0.60	2.92	0.122	1.74	2.84
S	1.09	0.29	0.010	0.10	0.11	0.82	0.032	0.46	2.87
N	17	18	14	18	12	12	12	18	18
79-027									
Surface									
$\bar{X}$	1.93	1.19	0.004	0.29	0.20	-	0.035	3.17	0.65
S	0.79	0.58	0.003	0.06	0.05	-	0.013	0.88	0.23
N	12	13	13	13	13	-	13	-	13
Bottom									
$\bar{X}$	1.34	0.65	0.003	0.28	0.18	-	0.043	1.86	2.04
S	0.58	0.29	0.002	0.05	0.03	-	0.017	0.49	0.65
N	21	22	21	22	22	-	21	-	13
80-018									
Surface									
$\bar{X}$	1.51	1.66	0.014	0.36	0.30	0.59	0.033	1.49	1.20
S	0.68	0.39	0.005	0.04	0.10	0.41	0.010	1.07	1.16
N	10	11	4	10	9	7	9	8	11
Bottom									
$\bar{X}$	1.27	1.34	0.010	0.34	0.24	0.80	0.030	1.29	2.41
S	0.59	0.69	0.004	0.04	0.10	0.88	0.010	0.70	1.88
N	10	11	5	11	11	8	9	10	10
81-010									
Surface									
$\bar{X}$	1.95	1.28	0.014	0.32	0.30	1.02	0.030	3.19	2.20
S	0.79	0.45	0.004	0.03	0.045	0.73	0.004	1.52	1.98
N	18	18	20	21	19	18	18	18	19
Bottom									
$\bar{X}$	1.22	0.71	0.011	0.32	0.29	-	0.030	1.78	6.28
S	0.31	0.22	0.002	0.02	0.05	-	0.004	0.53	3.43
N	5	6	6	6	6	-	18	6	7

results for all four cruises are similar. Higher average values were found for cruise 79-007 but this is probably due to the high input of fresh water runoff from the Saint John River that spring. In general trace metal results exhibit little spatial variability within the Bay of Fundy but the values are subject to changes due to enhanced fresh water inputs especially during the spring runoff.

3.2 Sediment Studies

Radionuclides released from the Point Lepreau N.G.S. will ultimately be partitioned between soluble and suspended particulate phases in proportions dependent on their respective particle-water distribution coefficients. The more particle-reactive radionuclides such as Co-60 and Zn-65 will follow the transport pathways for particulate matter through the water column and will ultimately be deposited in the sediments. Sites of net sediment deposition within the western Bay of Fundy warrant investigation because they may ultimately provide a historical record of the particle-reactive radionuclide releases from the Point Lepreau NGS. Further, knowledge of radionuclide sediment inventories is critical to the determination of the availability of these radionuclides for uptake by bottom dwelling organisms and to assess the importance of post-depositional remobilization of radionuclides by particle resuspension, bioturbation and diffusion. Finally, the sediment inventory of radionuclides may represent a significant fraction of the total budget for radionuclide releases from the Point Lepreau NGS which must be determined in order to estimate net radionuclide fluxes through different environmental phases.

3.2.1 Analytical Methodology

Sediment samples were collected at all stations during the AERU Bay of Fundy cruises using both Van Veen and Shipek grab samplers in order to identify fine-grained depositional regimes. Box cores and Lehigh gravity cores were collected subsequently at stations selected on the basis of the grab samples. The box cores were sampled at 1 cm intervals on board the research vessel (CSS Dawson) and the gravity cores were stored upright for transport to the laboratory where they were subsequently extruded and sub-sampled at 1 cm intervals. The water content of the samples was measured routinely as a function of sediment depth in order to determine the integrated accumulation of sediment solids in each core as a function of sediment depth.

Measurement of radionuclide activity in the dried sediments consisted initially of Pb-210, Ra-226, Cs-137 analyses but have recently been supplemented with Pu-238 and Pu-239,240 determinations. Pb-210 ($t_{1/2}$ = 22.3 yr) is a naturally-occurring member of the U-238 decay series which is introduced into coastal waters at a relatively constant rate (flux = 1.0 dpm/cm^2 yr) as a result of the radioactive decay of Rn-222 in the atmosphere. Pb-210 is rapidly scavenged by particulate matter in the water column and accumulates as an unsupported excess in the sediments. Measurements of the excess Pb-210 activity, $A_{Pb\text{-}210_{ex}}$ (equal to the total Pb-210 activity minus the supported Pb-210 activity due to radioactive decay of Ra-226 within the sediments), as a function of sediment depth can be used to determine sedimentation rates, rates of sediment mixing by bioturbation, or to identify anomalous depositional events associated with sediment slumping. Pb-210 analyses were performed by deposition of its granddaughter Po-210 and tracer Po-208 onto silver or nickel discs

from acid-digested sediment samples and Po-210 and Pb-210 activities were determined by alpha spectrometry following the methods outlined in Smith and Walton (1980). Radium-226 activities were determined both by gamma spectrometry and by the radon gas emanation method.

In contrast, Cs-137 ($t_{1/2}$ = 30 yr), Pu-238 ($t_{1/2}$ = 86.4 yr), Pu-239 ($t_{1/2}$ = 2.4 x 10^4 yr) and Pu-240 ($t_{1/2}$ = 6580 yr) were introduced initially into the environment during the 1950's as a result of nuclear weapons tests conducted in the atmosphere. Inputs of these radionuclides to the water column were maximized during the early 1960's, the period of most intensive weapons testing, and the timing of these weapons tests may be recorded as peaks in the activity profiles in undisturbed sediments. Cs and Pu measurements in sediment cores compliment Pb-210 studies of sediment transport processes and in addition provide baseline data which will be used to assess inputs from the Point Lepreau NGS. Cs-137 activities were determined by non-destructive analysis of dried sediment samples using Ge(Li) and hyper-pure Ge detectors interfaced with a multi-channel analyser.

The analytical procedures for Pu analysis have been developed recently in the AERU Laboratory to determine Pu-239,240 (the two plutonium isotopes cannot be resolved individually using the alpha-spectrometric counting technique employed) and Pu-238 in sediments. The techniques are based on commonly employed Pu methodologies (e.g. Talvitie 1971, 1972; Wong, 1971) and begin with the addition of a calibrated tracer as yield monitor (Pu-242) to a 10 g sample of dried sediment. After allowing 12 hr for isotopic equilibration, the Pu is leached from the sediment using a mixture of hot concentrated nitric (HNO_3) and hydrochloric (HCl) acids. The sediment is then removed by centrifugation and the process repeated.

After removal of remaining HCl, the solution is diluted to 8 M HNO_3 and passed through an anion-exchange column (Bio-Rad AG 1-X4, 100-200 mesh). Pu is absorbed as Pu IV complexes in high molarity HNO_3 and HCl while Am, Th, U, Po and other radionuclides liable to interfere with the Pu spectra and also Fe, Mn and other stable elements which can affect Pu electrodeposition are eluted. Pu isotopes are eluted from the column as Pu (III) using a dilute HCl/H_2O_2 eluant. A second, similar column ensures an efficient separation of Pu which is then electrodeposited on stainless steel discs and the activity of the nuclides measured by alpha spectrometry. For the current background levels of 0.0001 cpm in the region of interest in the alpha spectrum, a chemical yield of 70%, a counting efficiency of 30% and a counting time of 5000 minutes, the present detection limit for Pu-239,240 is approximately 0.0005 dpm/g of sediment (95% confidence limit).

The activity level of Pu-241 can also be measured through detection of its daughter product Am-241 after an ingrowth period of 1-2 years. A modification of the Pu methodology allows Am levels to be measured in sediment samples and will be applied after a suitable ingrowth period.

3.2.2 Surface sediments (0-1 cm)

Surface sediment studies of radionuclide activity can, in some cases, provide a quick method of detecting changes in the input of radionuclides to sediment (e.g. Assinder, 1982). This relies on the frequently observed, inverse relation between sediment particle size and radionuclide activity (e.g. Hetherington and Jefferies, 1974; Aston et al., 1981).

Classification of the surface sediments on the basis of their relative sand and mud content gives the particle size distribution pattern for

the western Bay of Fundy, indicated by Fig. 18 (Loring, 1979). The finest-grained sediments are found in the Quoddy region, west of Point Lepreau and extend in a lobe south to Grand Manan Island, generally consistent with the distribution pattern of LaHave clays (Fadar et al., 1977). This distribution of fine material has been verified by measurements of the percent mud (< 63 μm fraction) component on selected surface samples of the cores collected in this study. Particle size is the main factor controlling trace metal concentrations in the Bay of Fundy because the degree of adsorption of metals onto the detrital host mineral particles increases with the particle surface area. Therefore the finer sediments in the Quoddy region are a sink and an enrichment area for rare metals introduced from both natural and industrial sources such as the St. Croix drainage basin, coastal industries and offshore dredge dispersal sites (Loring, 1982).

Similar considerations apply to radionuclide activity distributions. The observed ranges and means ($\bar{x}$) for the various radionuclides (in dpm/g) and radionuclide ratios are as follows: $Pb\text{-}210_{ex}$ (2.1-11.4; $\bar{x}$ = 5.63), Cs-137 (0.07-0.51; $\bar{x}$ = 0.24), Pu-238 (0.001-0.005; $\bar{x}$ = 0.003), Pu-239,240 (0.03-0.15; $\bar{x}$ = 0.08), Pu-238/Pu-239,240 (0.028-0.042; $\bar{x}$ = 0.036), Pu-239,240/Cs-137 (0.22-0.65; $\bar{x}$ = 0.36), $Pb\text{-}210_{ex}$/Cs-137 (10.3-49.0; $\bar{x}$ = 26.5) and $Pb\text{-}210_{ex}$/Pu-239,240 (37.3-112.8; $\bar{x}$ = 75.8). These values are consistent with those previously noted in other nearshore N.W. Atlantic surface sediments (Livingston and Bowen, 1979). All the radionuclides have higher activities in the finer-grained sediments although an exact relationship to grain size is precluded by other factors such as (1) the variation of inputs with time which affects the amount of radionuclide available for uptake by a given grain-

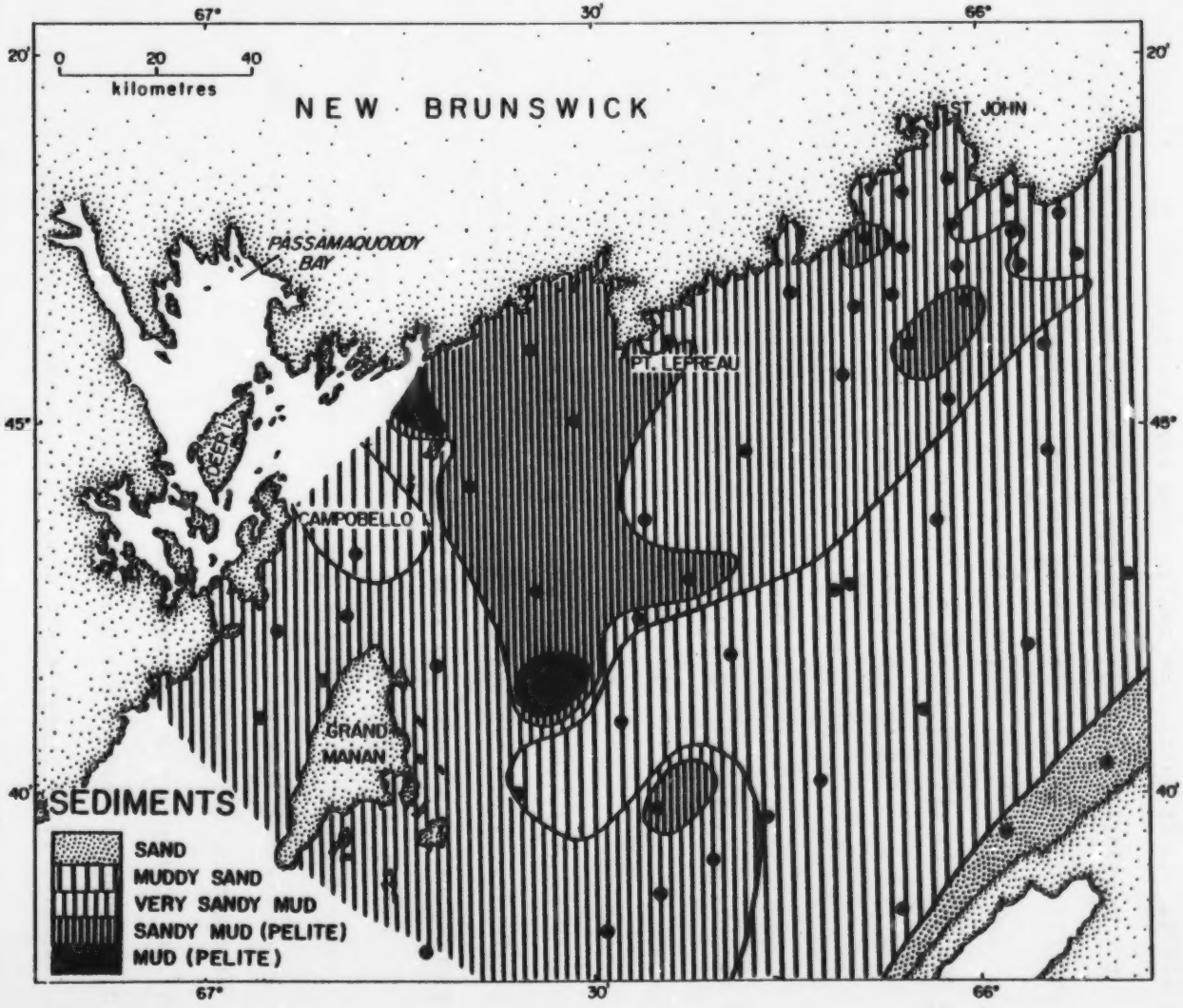


Figure 18. Particle size distribution of sediments in the western Bay of Fundy. Reproduced from Loring (1979).

size sediment; (2) the sedimentation rate which affects the period of input changes being sampled, and; (3) bioturbation and other sediment mixing processes. However, the measurement of sediment grain-size allows some degree of predictive capability for assessing additional radionuclide inputs to sediments from reactor releases.

3.2.3 Activity/Depth Profiles

The sediment depth profiles for excess Pb-210, derived apparent sedimentation rates and Cs-137 profiles have been discussed in detail in Smith et al. (1982) for cores from a variety of sites in the Bay of Fundy. In particular, a westerly transect through changing depositional regimes was used to illustrate the main conclusions from these measurements. Recent measurements on Pu-238 and Pu-239,240 activities in sediment samples collected on the same transect (Fig. 19) are discussed below.

Briefly, the highest inventories of excess Pb-210, $I_{Pb\text{-}210_{ex}}$, proportional to the total sediment flux of Pb-210 and the highest apparent sedimentation rates are found in the Quoddy region, near the northern approaches to Passamaquoddy Bay, where the sediments consist of fine-grained muds or pelites (Fig. 18). The fine-grained sediments in this region constitute a net depositional sink, both for Pb-210 and other particle-reactive radionuclides, in addition to trace metals as noted previously. In contrast, the sediments in the immediate vicinity of Point Lepreau which grade into sandy muds, are characterised by lower $I_{Pb\text{-}210_{ex}}$ and apparent sedimentation rates. This latter result is consistent with the inhibition of deposition of fine-grained material as the result of the strong residual bottom currents in this region. In addition, Cs-137 profiles indicate little sediment mixing in the higher sedimentation

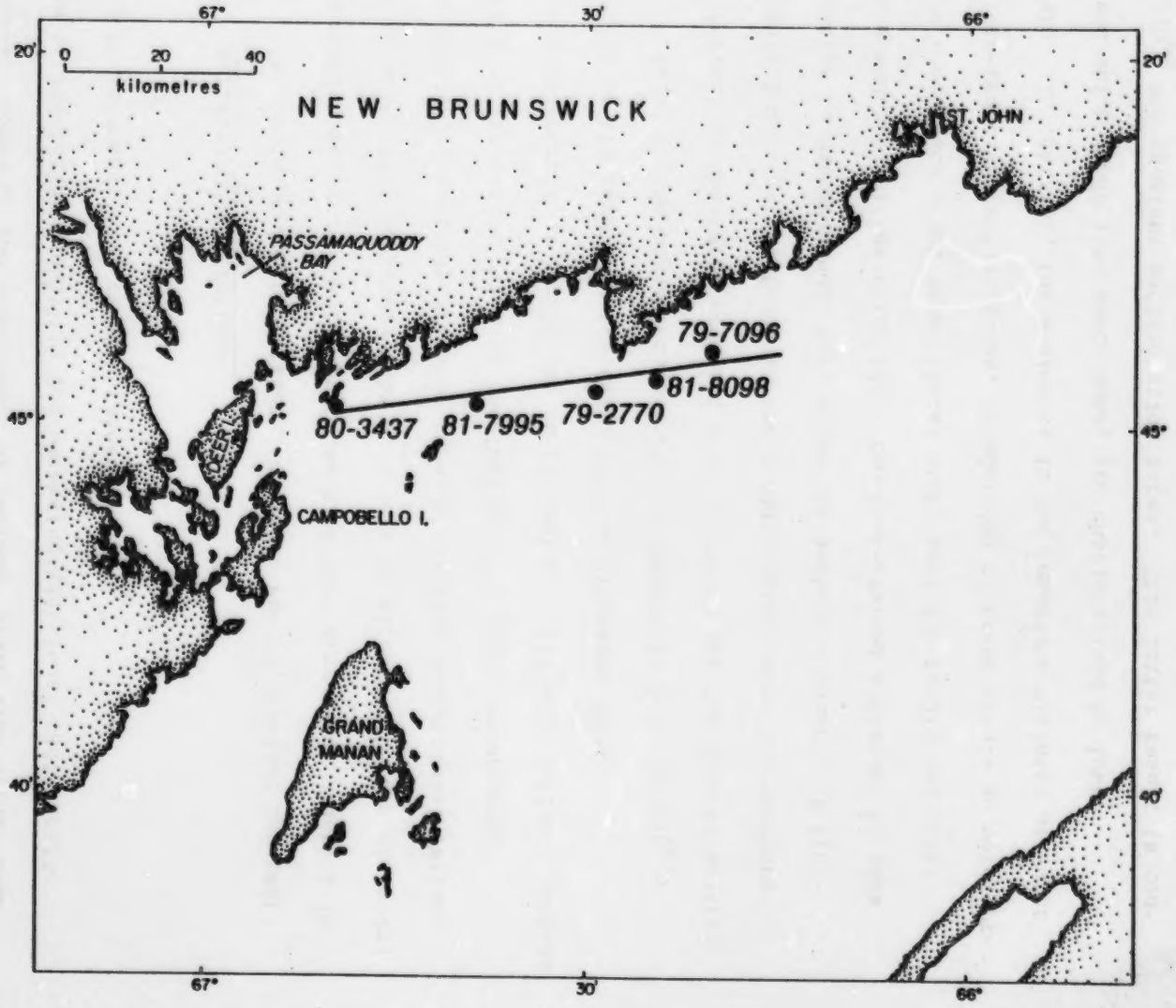


Figure 19. Core locations on westerly transect through several depositional regimes.

regimes but some evidence of mixing, probably as the result of bioturbation, in the Point Lepreau area (see below).

Pu depth profiles for cores 81-7995 and 81-8098 (Figs. 20, 21) illustrate the variations noted above for Pb-210 and Cs-137 and are representative of the changes in Pu profiles along the transect. The apparent sedimentation rate of 0.12 g/cm^2 yr for core 81-8098 characterises the low sedimentation rate/coarse-grained regime in the vicinity of Point Lepreau. In this core, Pu and Cs-137 are measurable at sediment depths corresponding to dates of deposition prior to 1952 (estimated from Pb-210 apparent sedimentation rates) which is the date for the initial introduction of significant quantities of these isotopes into the environment from nuclear weapons tests. Sediment mixing, probably as the result of bioturbation, is responsible for the excessive downward penetration of fallout radionuclides in these sediments. For this reason sedimentation rates estimated for these cores represent upper limits on actual sedimentation rates.

In contrast, core 81-7995 is representative of the higher sedimentation rate/fine-grained regime in the Quoddy region (apparent sedimentation rate = 0.61 g/cm^2 yr). The threshold for the initial appearance of both Cs-137 and Pu nuclides occurs near the sediment depth corresponding to a deposition date of 1952 estimated from the $Pb\text{-}210_{ex}$ profile. This good agreement between the Cs and Pu thresholds and the deposition date for their initial input into the worlds oceans indicates that there has been little sediment mixing in these regimes and that the apparent sedimentation rates may represent the actual sedimentation rate at these locations. It also suggests that there has been little post-depositional differential redistribution of Cs and Pu relative to either each

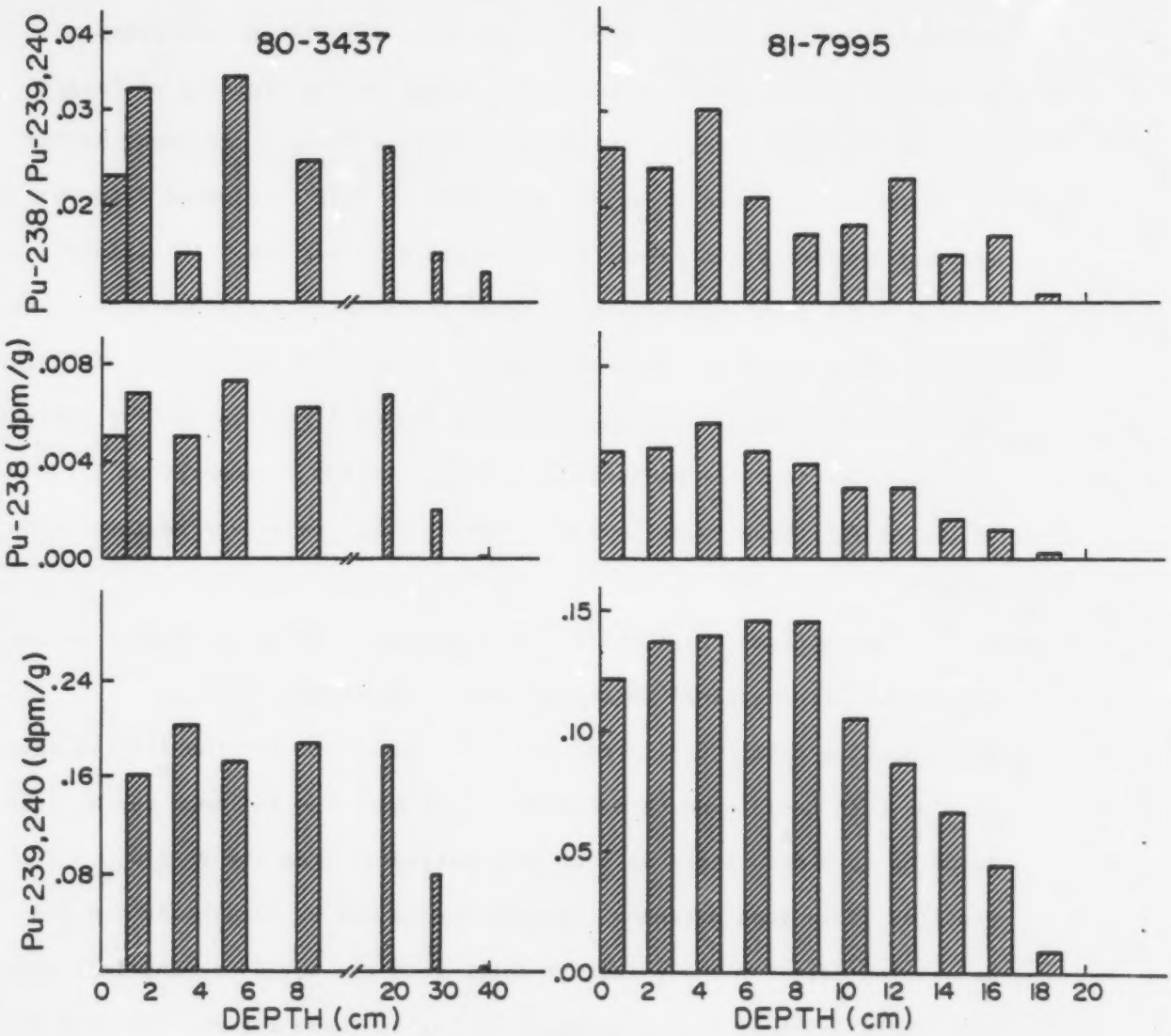


Figure 20. Plutonium activity and ratio profiles in cores 80-3437 and 81-7995.

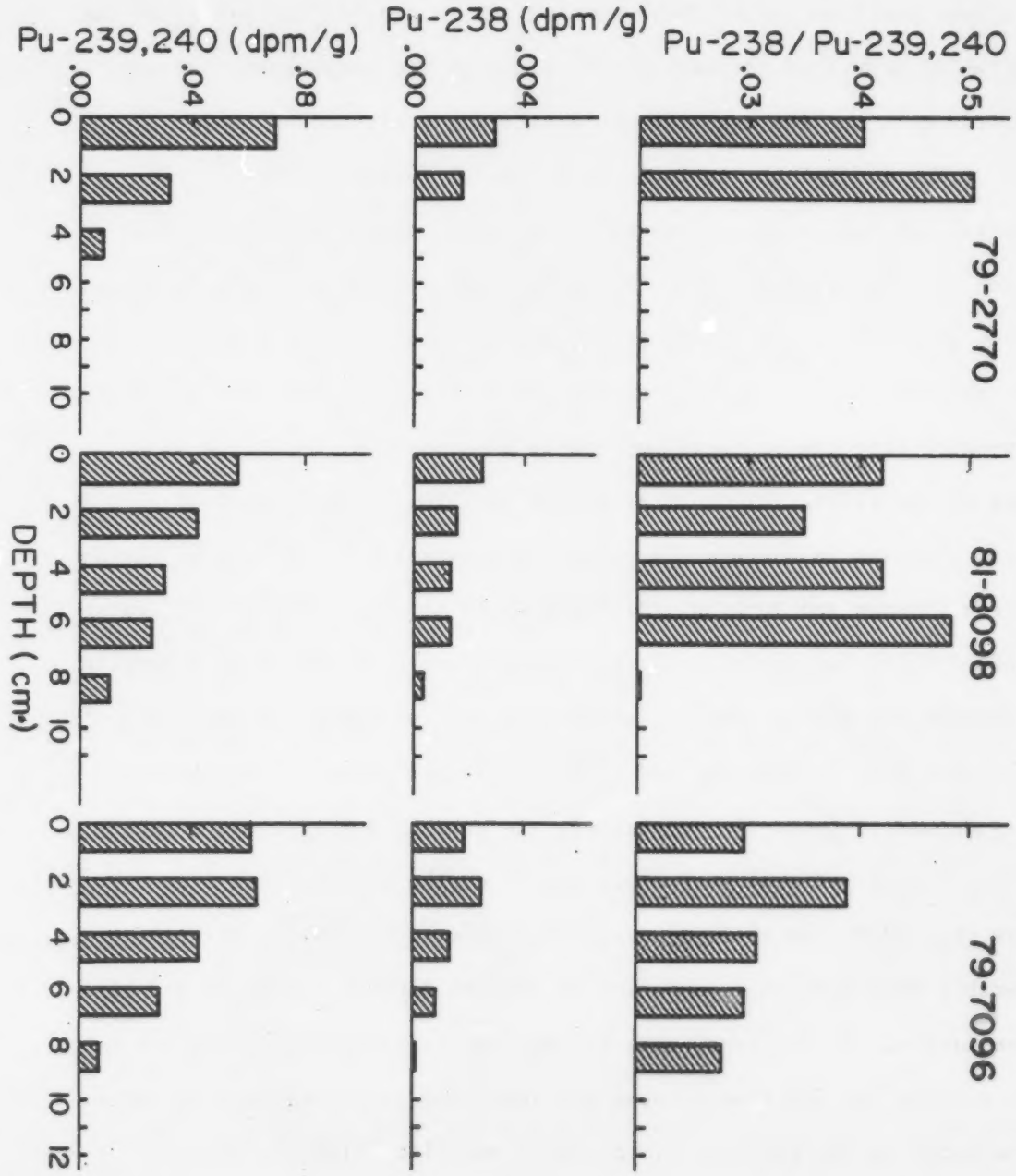


Figure 21. Plutonium activity and ratio profiles in cores 79-2770,81-8098 and 79-7096.

other, as has been suggested for some nearshore sedimentary regimes (e.g. Livingston and Bowen, 1979).

Some verification of these conclusions is provided by values of the Pu-238/Pu-239,240 ratio in these cores, although the compounded counting errors from both nuclide measurements limits the precision of these values. Prior to 1964 the isotopic ratio Pu-238/Pu-239,240 in nuclear weapons fallout debris was estimated to be 0.024, based on measurements conducted on a large number of soil samples collected in northern latitudes (Hardy et al., 1973). In April, 1964 a navigational satellite carrying a SNAP-9A thermal power generator, containing 17 kCi of Pu-238, was burned up upon re-entry into the southern hemisphere stratosphere, resulting in a tripling of the global inventory of Pu-238 by 1970. SNAP-9A-derived Pu-238 was first observed in surface air particulate material in 1966 at Richland, Washington (Thomas and Perkins, 1975) and since that time the isotope ratio Pu-238/Pu-239,240 has varied in different environmental phases in a complex manner (Smith and Ellis, 1982). In general, the isotope ratio of these radionuclides lies in the range of 0.03-0.15 in any sediments containing recently-deposited (post-1966) material. In core 81-8098, values of the Pu-238/Pu-239,240 ratio of >0.035 persist to depths greater than 7 cm (earlier than 1910 from $Pb\text{-}210_{ex}$ apparent sedimentation rates) indicating considerable mixing of some component of SNAP-9A derived Pu-238 in Bay of Fundy sediments. In contrast, core 81-7995 has values >0.030 above 7-8 cm (≈1971, showing the lag time between the event and its deposition in sediments as noted in the Saguenay Fjord (Smith and Ellis, 1982)), with a general decrease in values to a pre-SNAP level of ≈0.02 at 18-19 cm (≈1955). This emphasises the lower, though not necessarily the absence of, mixing in these regimes.

3.2.4 Radionuclide Inventories

Radionuclide inventories, inventory ratios and apparent sedimentation rates along the complete transect (estimated using bulk densities and mass depths for each core segment to correct for porosity and compaction) are plotted in Fig. 22 and tabulated in Table 10, in association with values from other studies for comparison. Apparent sedimentation rates and $I_{Pb\text{-}210_{ex}}$ trends have been discussed previously. Cs-137 inventories ($I_{Cs\text{-}137}$) and Pu-239,240 inventories ($I_{Pu\text{-}239,240}$) exhibit similar trends to Pb-210 inventories due to their mutual dependence upon the accumulation of fine-grained, high-activity sediments in an area of high apparent sedimentation rate. These inventories have comparable magnitudes to those observed in other nearshore Atlantic sediments (Table 10).

A mean fallout activity ratio of 0.012 was estimated by Harley (1975) for Pu-239,240/Cs-137 from measurements conducted on stratospheric particulate samples collected during the early 1960's. Pu-239,240/Cs-137 ratios in soils tend to be close to fallout debris ratios (Hardy et al., 1973), while elevated ratios (0.01-0.05) have been measured in freshwater sediments (Edgington and Robbins, 1975a; b). In contrast, Pu-239, 240/Cs-137 inventory ratios in Bay of Fundy sediments are greater by an order of magnitude than observed in other marine systems, (Livingston and Bowen, 1979) owing to reduced particle/water distribution coefficients (K_d) for Cs in higher ionic strength media such as seawater (Cs-137 must compete with seawater cations, especially potassium, for binding sites on particle surfaces). Pb-210/Cs-137 and Pb-210/Pu-239,240 inventory ratios are also similar to previously observed values in nearshore sediments (e.g. Carpenter and Beasley, 1981; Beasley et al., 1982) although there are few directly comparable values.

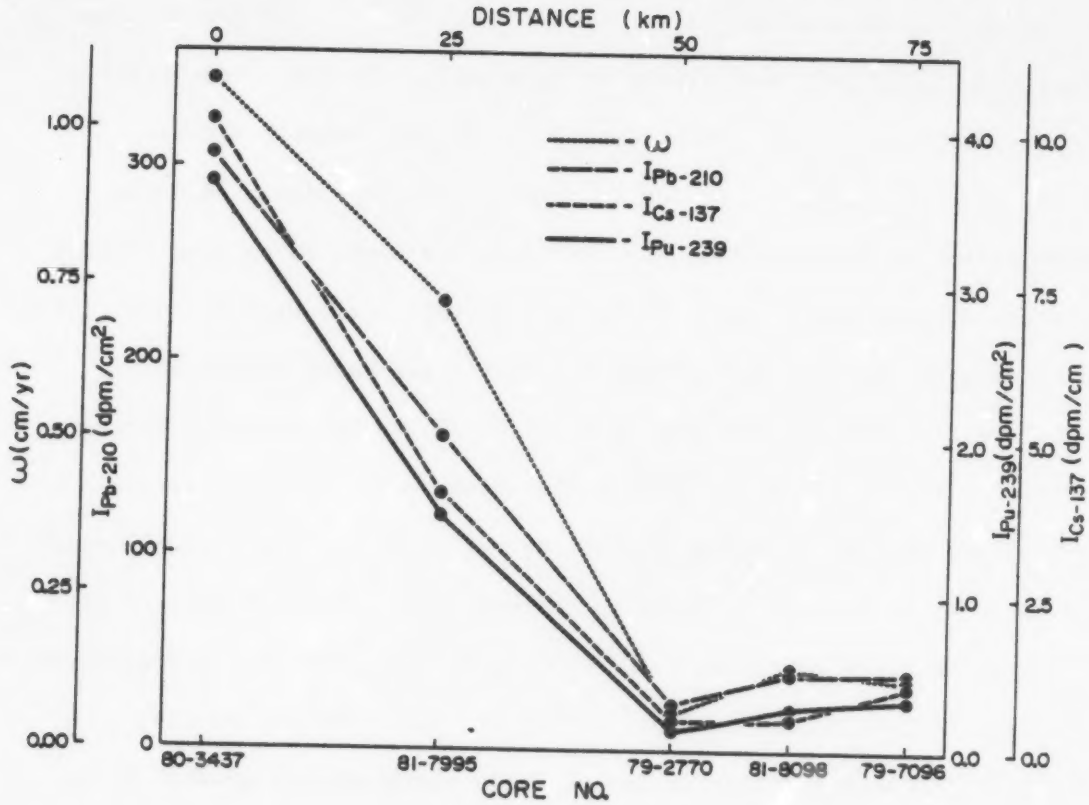


Figure 22. Sedimentation rates, Cs-137, Pu 239 and excess Pb-210 inventories for core location in Figure 19.

Table 10. Apparent sedimentation rates, radionuclide inventories and inventory ratios in Bay of Fundy and other nearshore sediments.

Core and/or Location	Apparent sedimentation rates		Inventories (dpm/cm^2)			Inventory ratios		
	cm/yr	g/cm^{2}yr	Pb-210$_{ex}$	Cs-137	Pu-239,240	Pb/Cs	Pb/Pu	Pu/Cs
Bay of Fundy								
80-3437	1.08	0.76	307	10.1	3.64	30.4	84.5	0.36
81-7995	0.72	0.61	161	4.10	1.40	39.3	115	0.34
79-2770	0.061	0.043	11.5	0.47	0.14	24.5	82.1	0.30
81-8098	0.13	0.091	18.9	0.50	0.21	37.8	90.0	0.42
79-7096	0.11	0.78	18.7	1.11	0.30	27.8	62.3	0.45
Range	----	----	11.5 - 307.4	0.47 - 10.1	0.14 - 3.64	24.5 - 39.3	62.3-115	0.30 - 0.45
Average	----	----	103.5	3.24	1.14	32.0	86.8	0.37
N.W. Atlantic (nearshore)(a)								
Range	----	----	----	0.87 - 3.26	0.24 - 1.00	----	----	0.15 - 0.39
Average	----	----	----	1.80 ± 0.75	0.47 ± 0.20	----	----	0.27 ± 0.08
N.E. Pacific (nearshore)(b)								
Range	----	----	39 - 72	----	0.33 - 0.73	----	92 - 117	----
Average	----	----	55 ± 16	----	0.53 ± 0.20	----	104 ± 12	----

(a) Livingston and Bowen (1979) (b) Carpenter and Beasley (1981)

Temporal and spatial changes in sedimentation, mixing and particle resuspension rates can produce different types of anomalies in the sediment-depth distribution of fallout and naturally-produced radionuclides. However, the total inventory of a radionuclide in the sediment column remains to a great extent, unaffected by these phenomena and the ratios of radionuclide inventories provide a convenient diagnostic for comparing different sedimentation regimes. The inventory ratios for Pb-210, Pu-239,240 and Cs-137 (Table 10) do not exhibit significant trends in this limited number of cores from the western Bay of Fundy despite the fact that they are characterized by sedimentation rates increasing by an order of magnitude on a transect between core 79-7096 and core 80-3437. The overall inventory of each radionuclide increases approximately linearly with the sedimentation rate suggesting that the same class of particle size distributions may be responsible for radionuclide scavenging and deposition in all of the sedimentation regimes.

Particle/water distribution coefficients for metals and radionuclides (K_d = activity in particulate phase (dpm/g)/activity in soluble phase (dpm/g)) vary with sediment grain size in a complex manner although, in general, the K_d for Pu and Pb is greater than that for Cs (Krishnaswami and Somayajulu, 1975; Murray and Eicke, 1977). Non-linear changes in K_d with varying grain size could produce changes in radionuclide inventories over the broad range of grain sizes which characterize sediments in the western Bay of Fundy. Significant changes in the organic matter composition of the sediments could also result in relative changes in radionuclide inventories, owing to the different rates of association of each radionuclide with organic material in the water column. Post-depositional sediment resuspension and bioturbation processes are additional

factors which might result in different inventories of these radionuclides in low sedimentation compared to high sedimentation rate regimes. However, the comparative uniformity of ratios in radionuclide inventories indicates that particle transport pathways for the different radionuclides may be remarkably similar and that some of the processes noted above may be of minor importance in Fundy sediment regimes. Additional analyses of sediment cores from this region are required to adequately characterize the spatial distribution of radionuclide inventories.

The factors affecting the variability of radionuclide inventories and ratios in the Bay of Fundy is a complex problem but an important one for the future consideration of particle-reactive radionuclide inputs from the Point Lepreau reactor.

4.0 ATMOSPHERIC MONITORING

The atmosphere is the most dynamic environmental medium for the transport of gaseous and particulate emissions from nuclear power facilities. The dispersion of radioactive isotopes released via stack gases follows several pathways: direct deposition, transport via air movement and subsequent exchange with surface water and transfer to various media via precipitation of airborne particulates.

The two principal components of the atmospheric monitoring program at Point Lepreau are (1) the compilation of a meteorological data base for the Point Lepreau region which can be applied in projections of the transport of airborne pollutants under diverse weather conditions and in atmospheric transport models and (2) the collection of atmospheric water vapour and particulate samples for radionuclide analysis.

4.1 Meteorological Data Collection

The meteorological station established in 1978, at Hall's Lake Siding (Fig. 23, site #1 approximately 4 miles north-northeast of the Point Lepreau NGS) has been in service for the last 4 years.The station consists of; (1) U2A wind equipment to measure wind speed and direction 10 metres above ground level, (2) a Fisher and Porter precipitation gauge and (3) a thermohydrograph to record temperature and relative humidity. Hall's Lake Siding was chosen as a meteorological station for several reasons, (1) lack of obstructions which could generate random noise in the meteorological measurements and (2) accessibility for routine maintenance.

The meteorological data collected at Hall's Lake Siding provides important information for characterizing the meteorological conditions prevalent in the Point Lepreau area. Tables 11 and 12 summarize the wind

Figure 23. Location of the four air monitoring stations in the vicinity of Point Lepreau. A fifth station is located at Digby, N.S.

Table 11. Percent frequency distributions of windspeeds and direction from U2A Wind Sensor at Hall's Lake Siding during period March 1979 to August 1981. Windspeeds in knots.

Direction	Period June–Sept.		Period Dec. Jan. Feb. Mar.		Annual	
	% Frequency	Mean Speed	% Frequency	Mean Speed	% Frequency	Mean Speed
N	1.7	4.4	4.1	8.4	2.8	6.3
NNE	1.4	4.5	3.5	6.1	2.5	5.9
NE	1.2	2.5	2.1	3.2	1.6	3.2
ENE	1.5	2.8	2.1	4.1	1.8	3.4
E	3.0	2.5	7.3	7.1	5.6	5.3
ESE	3.7	3.8	6.2	6.1	5.5	4.8
SE	3.6	4.0	6.2	4.6	4.6	4.4
SSE	6.6	4.2	3.7	7.6	5.8	4.8
S	14.9	4.6	7.9	7.3	12.1	5.4
SSW	21.9	4.3	8.6	7.0	16.0	5.0
SW	21.0	4.1	10.6	6.2	15.7	4.7
WSW	5.4	4.2	11.2	6.6	7.0	5.5
W	4.8	5.0	9.6	6.6	6.6	5.6
WNW	3.0	6.1	6.6	6.9	5.0	6.9
NW	3.6	7.1	7.6	8.8	4.9	8.5
NNW	2.4	6.0	2.5	7.4	2.2	6.7

Table 12. Percent frequency and mean wind speed by direction at Hall's Lake Siding, N.B. During the period September 1981 through August 1982.

Direction	Period Sept.'81 - Jan.'82		Period April - August '82.		Annual	
	% Frequency	Mean Speed	% Frequency	Mean Speed	% Frequency	Mean Speed
N	6.4	8.2	7.5	9.0	7.0	8.6
NNE	8.6	9.1	5.3	7.9	6.8	8.5
NE	8.9	7.0	3.3	3.9	5.9	5.5
ENE	3.5	6.1	1.3	3.9	2.3	5.0
E	1.7	4.8	3.6	4.1	2.7	4.5
ESE	2.4	8.9	4.3	4.3	3.4	6.6
SE	1.9	7.6	3.9	5.2	3.0	6.4
SSE	1.3	8.4	4.8	6.5	3.2	7.5
S	4.0	8.7	6.6	6.1	5.4	7.4
SSW	5.0	7.8	13.1	6.4	9.4	7.1
SW	6.3	6.9	11.9	6.8	9.3	6.9
WSW	3.9	7.2	5.5	6.1	4.8	6.7
W	4.6	7.4	4.9	6.7	4.8	7.1
WNW	2.9	8.7	4.7	7.5	3.9	8.1
NW	4.0	6.6	3.8	7.3	3.9	7.0
NNW	5.2	8.7	2.6	7.7	3.8	8.2
Calm	29.4		12.9		20.4	

Notes: (1) Wind Speed is in knots

(2) February and March 1982 are not included due to missing data.

speed and direction data for 1981 and 1982. These results confirm the marked preference for a southwesterly wind direction, more pronounced during the warmer, summer months (Smith et al., 1982). The mini-sonde experiment carried out in 1980 (Smith et al., 1981) indicated that the summer months were characterized by low atmospheric mixing and low ventilation coefficients. This coupled with the prevailing southwesterly winds suggests a strong probability that airborne materials originating from the Point Lepreau NGS will be transported inland on a northeasterly transect.

The winter air circulation patterns are characterized by a strong northeasterly component. The interaction of this cooler air blending with the warmer Bay of Fundy air masses generally results in a high degree of mixing vertically as well as laterally. In the event of radionuclide releases in the winter months, it is expected that airborne particles from the Point Lepreau NGS will be dispersed more efficiently in the atmosphere as they cross the Bay of Fundy.

Rainfall data has been collected from both Hall's Lake Siding and Saint John (Table 13) in an effort to compare the meteorological conditions at both sites. Rainfall for 1981 and 1982 show some variation from month to month with monthly totals generally being higher at Saint John. Year-end figures indicate that there was 14% more rain in both 1981 and 1982 at Saint John than at the Musquash station. The results of both the wind speed and rainfall data indicate that the Saint John airport data is indicative of meteorological events in the Point Lepreau area but is not an adequate substitute for a meteorological station situated in the immediate vicinity of the NGS.

Table 13. Comparison in inches of monthly precipitation collected at stations located in Saint John and Musquash, New Brunswick.

	1981		1982	
Month	Saint John	Musquash	Saint John	Musquash
January	5.28	3.20	- *	-
February	5.16	5.60	-	-
March	3.98	1.80	4.53	4.10
April	5.98	6.00	5.83	4.20
May	5.83	5.00	1.61	1.10
June	8.50	6.40	5.98	6.00
July	-	-	5.00	3.60
August	-	-	4.96	3.70
September	4.25	4.80	3.94	3.00
October	7.91	7.42	1.73	2.10
November	5.43	5.00	6.34	5.50
December	-	-	4.45	3.40
Yearly Totals	52.32	45.22	44.37	36.70

* - Indicates no data available for that month.

4.2 Air Monitoring Stations

Four air monitoring stations, described in Bishop et al. (1980) and illustrated schematically in Fig. 24, have been deployed in the Lepreau region at the sites noted in Fig. 23. Three of these stations are operated by local residents and the fourth by the New Brunswick Electric Power Commission's Nuclear Operations Group. Two of the stations (Point Lepreau and Dipper Harbour) are located near similar stations owned and operated by NBEPC, allowing for a comparison of data. A fifth station was established at Digby, N.S. to assess the magnitude of airborne radionuclide transport from Point Lepreau across the Bay of Fundy.

During the pre-operational period, the air monitoring stations are operated for approximately one week each month. Air is drawn continuously into the system at a height of 1 metre above the ground via steel piping that extends vertically through the bottom of the sampler housing. Thus, the collection of wet and dry precipitation is minimized except under the most adverse meteorological conditions.

The one quarter horsepower, oiless vacuum pump (Gast Manufacturing Corp.) draws air through a 47 mm diameter type "A" glass fibre filter and a cartridge of activated coconut shell carbon. The filter removes 0.3 μm size particles from the air stream with a retention efficiency of 99% and the charcoal bed absorbs elemental and organically-bound forms of iodine with a retention efficiency of 93%. The glass fibre filter rests on the O-ring with good separation from the pre-filter and air is drawn through the cap of the charcoal cartridge. Flow rates, determined by timing the passage of a known volume of air through the integrating flow meter, drops during sampling because of particulate accumulation and subsequent filter clogging. This drop in the flow rate often results in a lowering of the

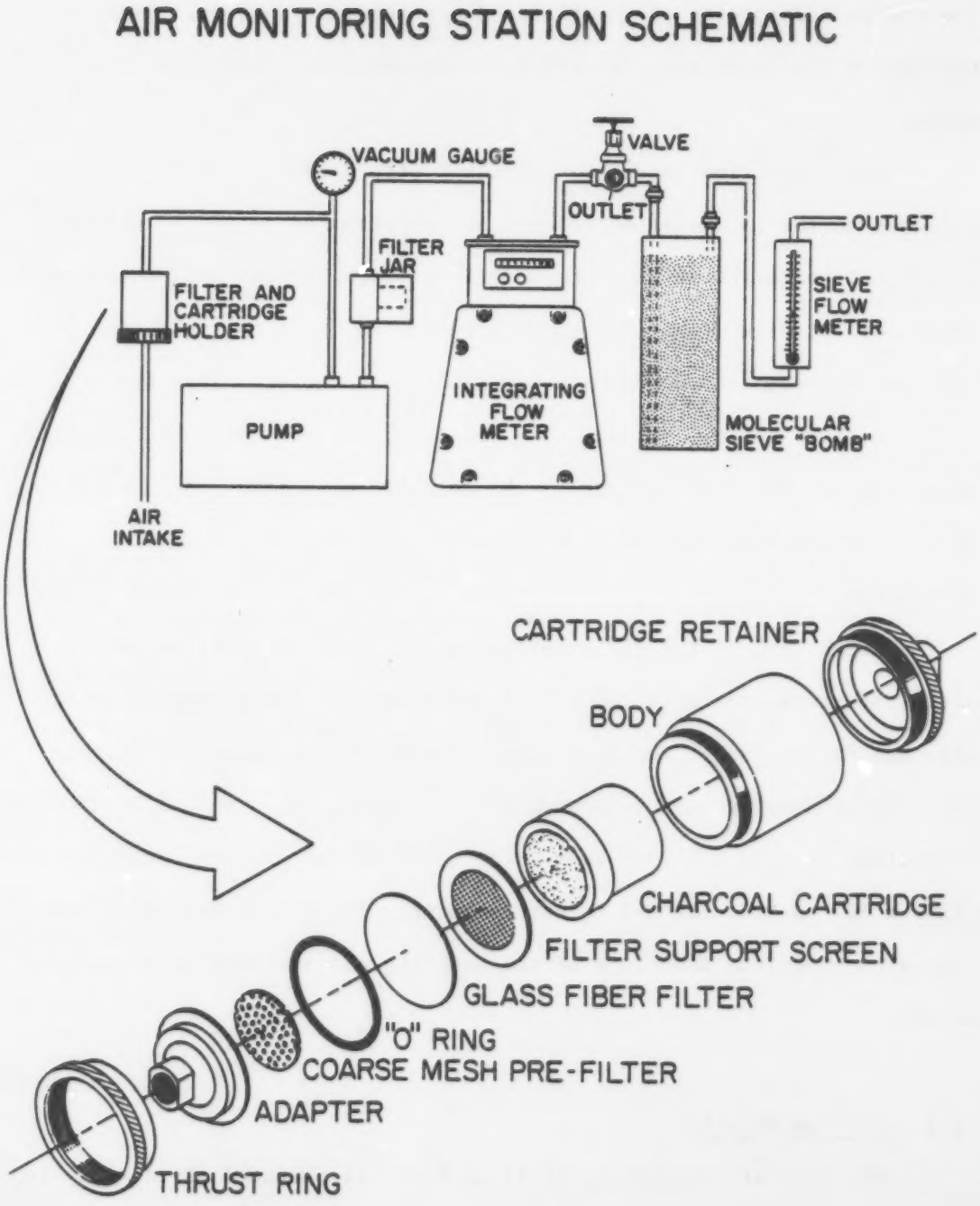


Figure 24. Schematic diagram of air monitoring station used in the field to collect water vapour (on molecular sieve),particulates (glass fiber filter), and I-131 (on charcoal cartridge).

flow rate through the molecular sieve bomb, used to remove water vapour from a diverted portion of the filtered air stream. However, the data supplied by the operators, is used to compensate for these flow rate changes.

Sieve air flow rates are regulated with a calibrated flow meter connected to the cannister outlet and are adjusted according to estimates of seasonal variation in absolute humidity. This ensures that the water sample collected is within the sieve's retention capacity. Relative humidity and temperature data are available for Station 1 from the meteorological station deployed at the same location, and the absolute humidities during the air sampling periods may be calculated. These are compared with sieve water recovery data as an additional check on the system's performance.

At the end of the air sampling period, the operators return the particle filters, charcoal cartridges and molecular sieve cannisters for radionuclide analyses. The approximate detection limits, which vary depending on sampling and instrumental conditions, are 4×10^{-5} Bq m^{-3} for particulate Cs-137, 3×10^{-5} Bq m^{-3} for I-131 on the charcoal cartridge and 1 to 7×10^{-2} Bq m^{-3} for H-3 in water vapour. These LD's assume a decay interval between the midpoints of the sampling and analysis intervals of ten days.

4.2.1 Tritium Results

The five air monitoring stations have all operated efficiently in 1982. The individual sampling periods have ranged from 6 to 8 days, during which time 400 to 1110 cubic meters of air were drawn through the filters. The variation in sample volume results from varying particulate levels in

air and varying sample periods. The molecular sieve cannisters were returned to the AERU laboratory from the field where water vapour was removed from the molecular sieve material by heating under atmospheric pressure at 350°C. The distillate was collected in a refrigerated trap held at -20°C and was isolated from laboratory air by a moisture trap at the system outlet.

The results of tritium analysis of atmospheric moisture are given in Fig. 25. The error bars represent the two sigma counting uncertainty. The total volume of air passing through the molecular sieves and the total weight of atmospheric water vapour collected are measured for each sampling run and this allows the calculation of tritium activities in terms of air volume, and the integrated absolute humidity for each sampling period. Tritium activity calculated for air volume is plotted against sampling midpoint in Fig. 26 and clearly shows a maximum in the summer months. A comparison of Fig. 26 and Fig. 27 which shows sample humidity compared to theoretical absolute humidities clearly indicates that the summer peak in tritium activity is directly related to an increase in humidity at that time of year.

The results for tritium in atmospheric moisture for 1982 are similar to the 1981 results and indicate that tritium is transported in the study area with water vapour and that there is no apparent source term for tritium that operates independently of the continent-ocean-atmosphere hydrological cycle. Thus, there is no evidence from these results for enrichment due to stratospheric inputs of tritium to the atmosphere due to recent nuclear weapons tests. The mean tritium activity measured in 1982 was 6 ± 2 Bq/l of air moisture. The absence of a seasonal bias in baseline levels of tritium in atmospheric water vapour should simplify the procedure

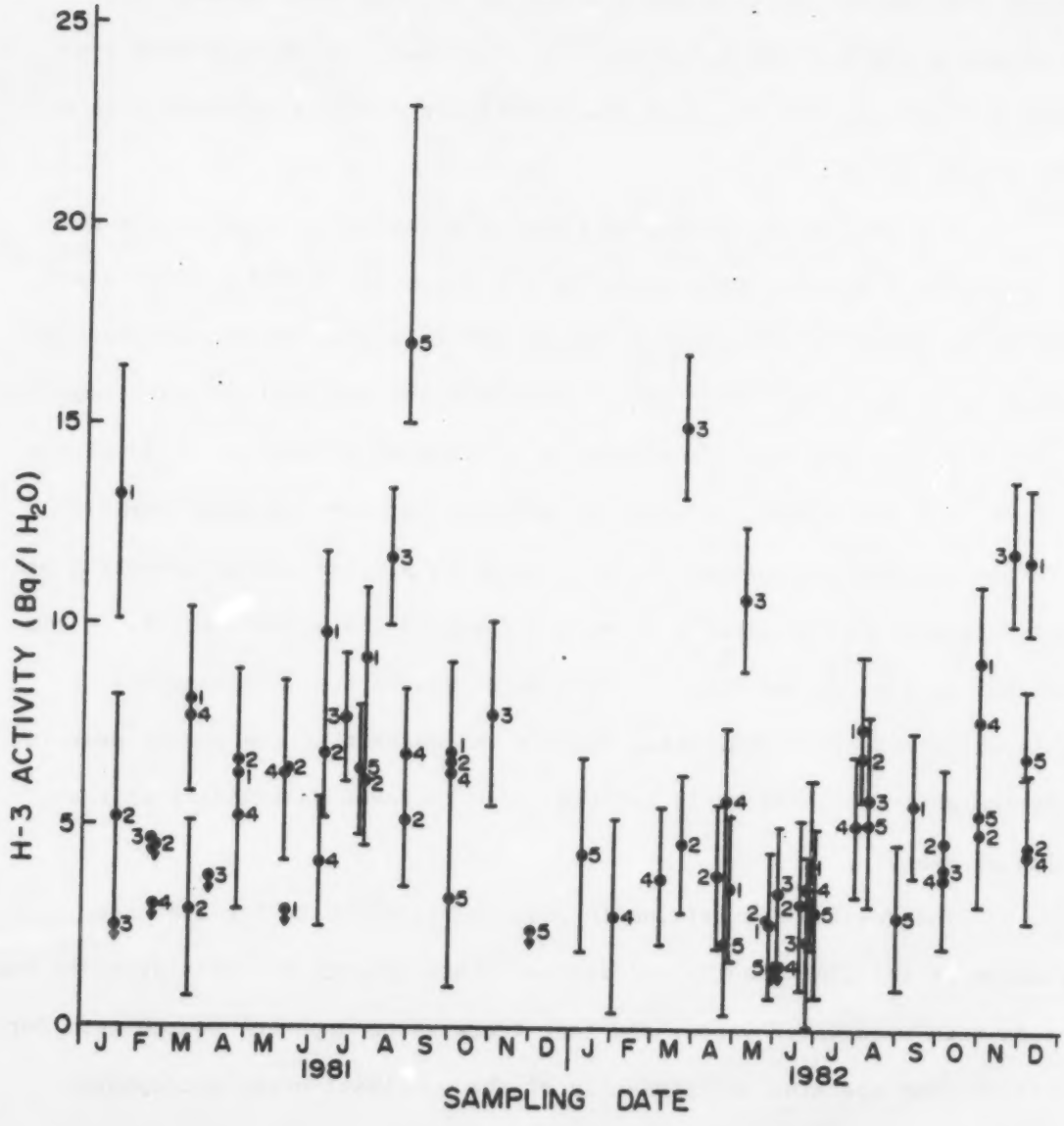


Figure 25. Tritium measurements (Bq/L) for different air monitoring stations (station numbers are illustrated in Figure 24) as a function of sampling date; arrows indicate values below the detection limit.

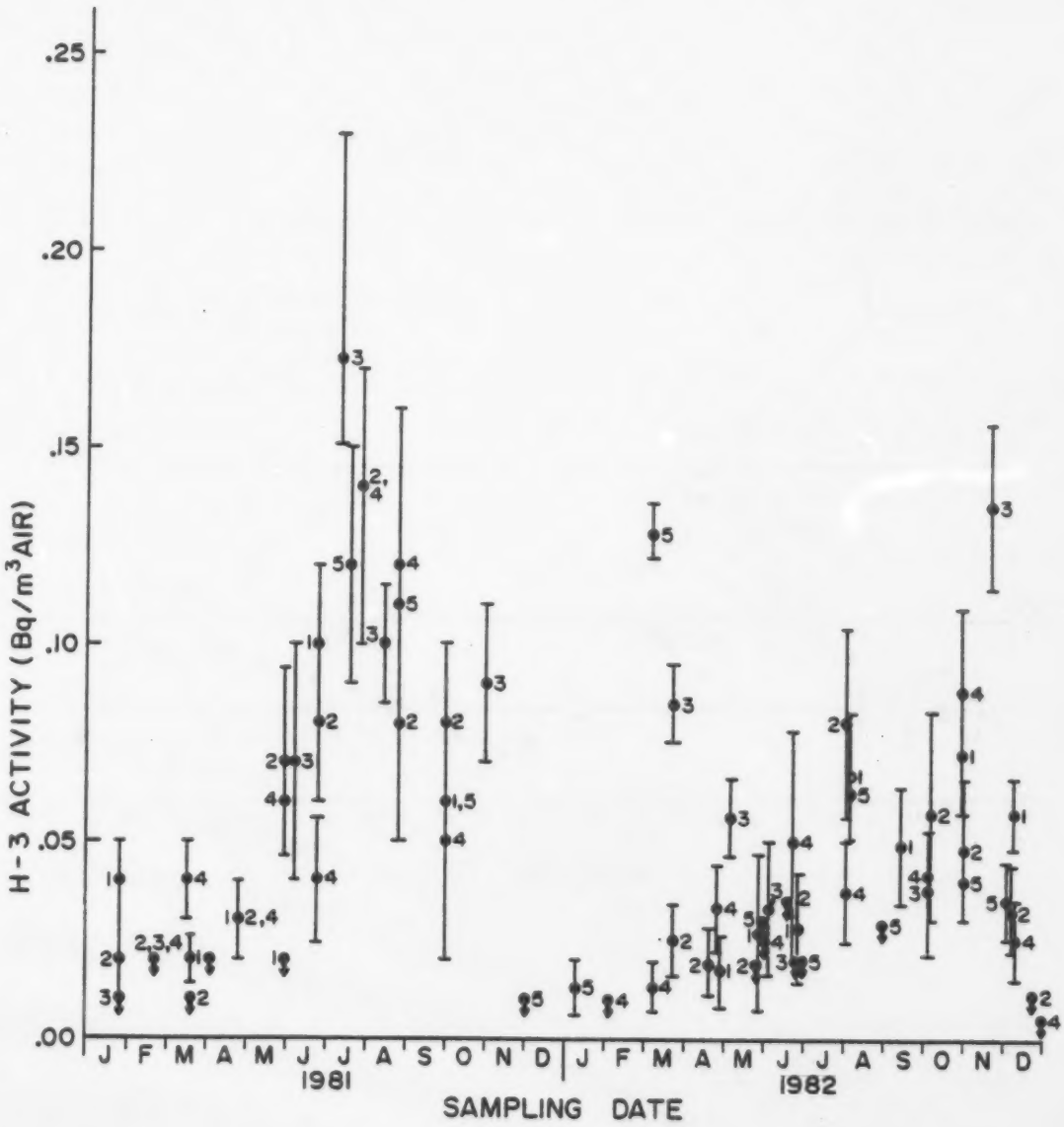


Figure 26. Tritium activities as a function of air volume sampled for different sampling intervals. Summer peak in tritium is caused by enhanced atmospheric moisture content.

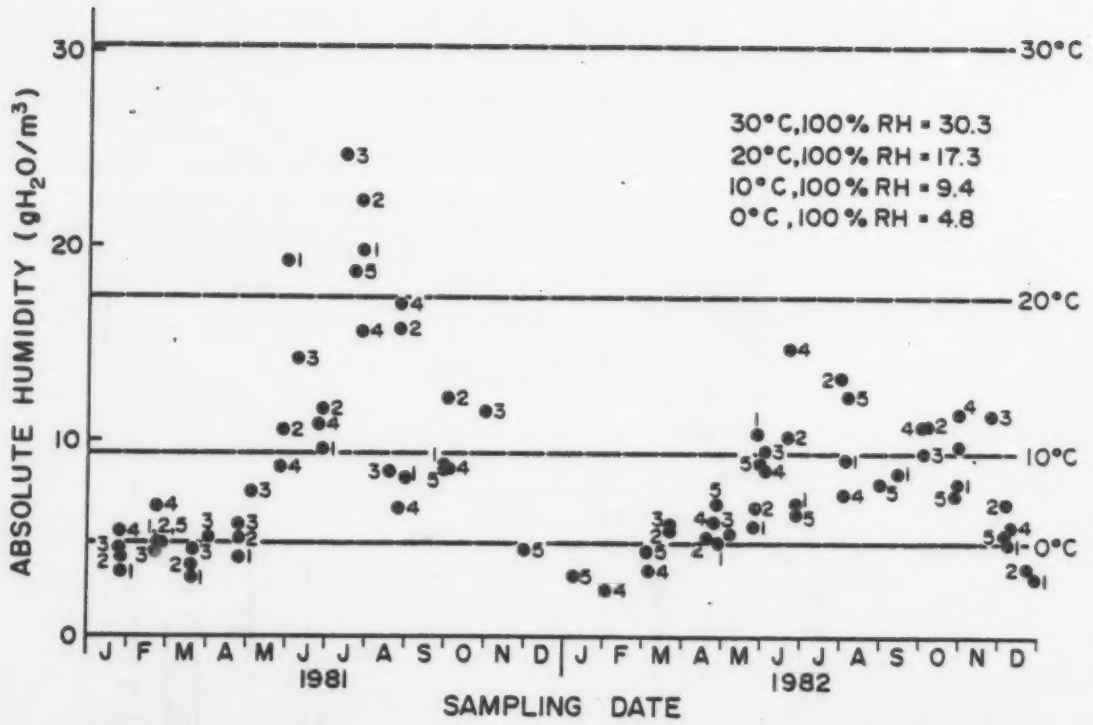


Figure 27. Absolute humidity for the sampling periods corresponding to tritium measurements given in previous figures. Dashed lines correspond to theoretical absolute humidities at the different temperatures.

for identifying releases of tritium from the Point Lepreau NGS.

4.2.2 Radionuclides in Atmospheric Particulates

Air filters containing airborne particulate matter were analysed for gamma-emitting radionuclides on hyper-pure Ge or Ge(Li) detectors. Results for air filters from 1982 are presented in Table 14. The station numbers correspond to the locations shown in Fig. 23. The sampling date refers to the sampling midpoint. With the exception of Be-7, which is a cosmogenic tracer, no radionuclides were measured above detection limits in 1982 (Table 14). Fission products were measured on air particulate material in late 1980 and 1981 as a result of the pulsed input of radionuclides into the atmosphere produced during the nuclear test carried out on October 16, 1980 by the People's Republic of China. Levels of fallout products in air particulate material fell below the detection limit by September 1981. This event provided an excellent field test for the AERU air monitoring program (Smith et al., 1981).

As shown in Fig. 28 and Table 14, by the end of 1981 the radionuclide levels were returning to pre-test values. The results for samples collected in 1982 indicate that all fallout radionuclide levels had fallen below their detection limits in air particulate material.

4.2.2.1 Beryllium-7

Beryllium-7 is a naturally occurring radionuclide which is the product of the cosmic ray fragmentation of atmospheric nitrogen and oxygen which takes place in the upper stratosphere. In the case of Be-7 for which there is no significant latitude dependence, the atmospheric distribution depends on the source region and transport mechanisms in effect. Its high

Table 14. Radionuclide results for air particulate material collected on air filters at air monitoring stations identified by location number in Figure 20.

Sample No.	Station	Date	Volume (m^3)	Radionuclide Activities (mBq/m^3) Cs-137	Be-7 ± 2σ
82-00650	1	4/02/82	744.3	<0.11	3.7 ± 0.7
82-00651	3	23/03/82	699.4	<0.06	2.7 ± 0.5
82-00652	4	1/02/82	996.8	<0.06	3.8 ± 0.4
82-00653	5	8/01/82	842.7	<0.04	2.4 ± 0.3
82-00654	1	5/03/82	882.5	<0.04	2.8 ± 0.3
82-00657	4	6/03/82	852.3	<0.04	2.8 ± 0.3
82-00658	5	6/03/82	679.0	<0.05	2.5 ± 0.4
82-00659	1	27/04/82	832.5	<0.05	4.1 ± 0.4
82-00660	2	19/04/82	730.6	<0.04	6.5 ± 0.4
82-00661	3	5/05/82	812.7	<0.05	0.9 ± 0.4
82-00662	4	24/04/82	818.4	<0.04	4.1 ± 0.4
82-00663	5	25/04/82	986.0	<0.04	1.1 ± 0.4
82-09901	1	28/05/82	792.9	<0.04	2.4 ± 0.4
82-09902	2	25/05/82	699.4	<0.05	4.6 ± 0.4
82-09903	3	3/06/82	623.0	<0.07	2.5 ± 0.4
82-09904	4	1/06/82	764.6	<0.05	2.6 ± 0.4
82-09905	5	30/05/82	760.6	<0.05	2.6 ± 0.4
82-09906	1	27/06/82	886.3	-	-
82-09907	2	19/06/82	679.6	<0.05	1.6 ± 0.4
82-09908	3	24/06/82	623.0	<0.02	2.0 ± 0.2
82-09909	4	22/06/82	764.6	<0.06	1.8 ± 0.6
82-09910	5	28/06/82	929.9	<0.04	1.9 ± 0.4
82-09911	1	2/08/82	781.5	<0.05	2.3 ± 0.5
82-09912	2	30/07/82	433.2	<0.09	<0.94
82-09913	3	5/08/82	812.7	<0.05	2.4 ± 0.5
82-09914	4	30/07/82	730.6	<0.06	1.4 ± 0.6
82-09915	5	6/08/82	437.8	<0.09	3.6 ± 0.2
82-09916	1	11/09/82	-		-
82-09917	2	1/10/82	657.0	<0.06	0.7 ± 0.5
82-09918	3	30/09/82	790.0	<0.05	1.5 ± 0.5
82-09919	4	1/10/82	506.9	<0.08	1.3 ± 0.7
82-09920	5	29/08/82	792.3	<0.09	1.6 ± 0.7
82-09921	1	26/10/82	909.0	<0.02	1.1 ± 0.3
82-09922	2	26/10/82	707.9	<0.06	2.3 ± 0.6
82-09923	3	19/11/82	600.3	<0.06	2.7 ± 0.6
82-09924	4	27/10/82	707.9	<0.05	1.6 ± 0.5
82-09925	5	26/10/82	967.0	<0.04	2.8 ± 0.4
82-09926	1	4/12/82	792.9	<0.04	2.3 ± 0.4
82-09927	2	3/12/82	875.0	<0.04	< 0.4
82-09929	4	6/12/82	744.9	<0.05	2.0 ± 0.5
82-09930	5	30/11/82	826.7	<0.04	2.9 ± 0.4
82-09931	1	23/12/82	829.7	<0.07	2.4 ± 0.6
82-09932	2	24/12/82	849.7	<0.04	0.8 ± 0.4
82-09933	3	26/12/82	1478.1	<0.02	1.7 ± 0.3
82-09934	4	25/12/82	1537.6	<0.02	1.1 ± 0.2
82-09935	5	28/12/82	879.4	<0.04	2.1 ± 0.4

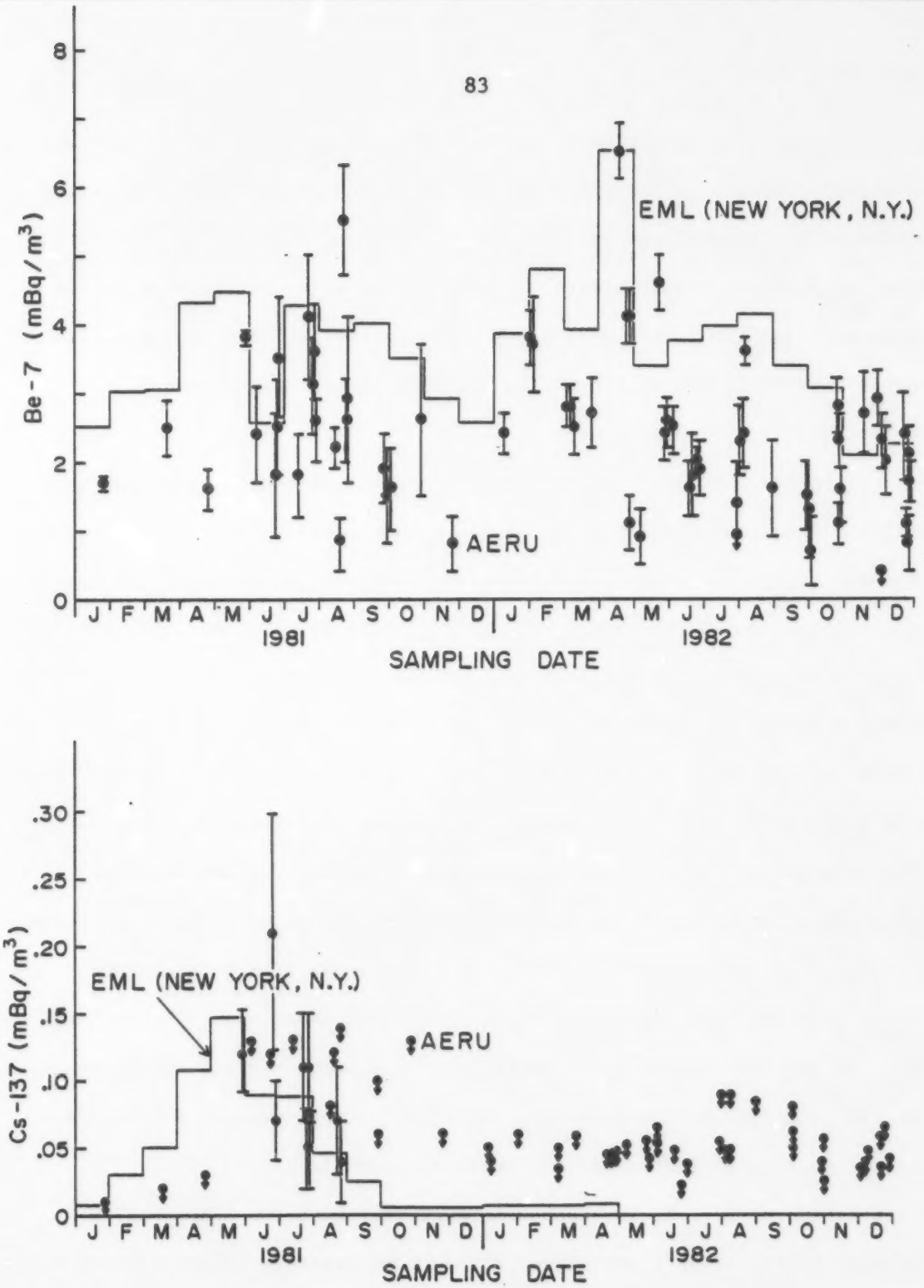

Be-7 (mBq/m³)
EML (NEW YORK, N.Y.)
AERU
J F M A M J J A S O N D J F M A M J J A S O N D
1981
1982
SAMPLING DATE
Cs-137 (mBq/m³)
EML (NEW YORK, N.Y.)
AERU
J F M A M J J A S O N D J F M A M J J A S O N D
1981
1982
SAMPLING DATE

Figure 28. (Upper figure) Be-7 activity of air particulate material for 1980-1982. Arrows correspond to values below the detection limit, crosses indicate values measured by Environmental Measurements Laboratory (EML) at New York, N.Y. (Lower figure) Cs-137 activity of air particulate matterial collected near Point Lepreau. Enhanced values in summer of 1981 due to Chinese nuclear weapons test, in October of 1980.

specific activity, resulting from its short half life (53 days), the lack of dilution by stable beryllium and its particle affinity have led to the extensive use of Be-7 for estimating the residence time of aerosols in the upper troposphere. Furthermore, because of its wide distribution in the atmosphere and its particle-reactive characteristics, Be-7 has proven useful as a particle tracer for rapid processes in the marine and aquatic environment (Krishnaswami, et al., 1980).

In the past, the input of Be-7 from nuclear testing has been undetectable in ground level air (Gustafson et al., 1961). More recently, Be-7 from the testing of French nuclear devices has been detected in the stratosphere (Sagar and Goel, 1980). The Be-7 results for 1981 and 1982 are presented in Fig. 28. The range of Be-7 values measured in 1981 is 0.8 ± 0.4 to 5.5 ± 0.8 mBq/m^3, with an average value of 2.5 mBq/m^3. This is similar to the range for 1982 values of 0.7 ± 0.5 to 6.5 ± 0.4 mBq/m^3, with an average Be-7 value of 2.4 mBq/m^3, indicating that there is minor variation in mean Be-7 levels from year to year. The peak in Be-7 activity, illustrated in Fig. 28, that occurs during the spring and summer, has been attributed to the scavenging effects of rain rather than to a seasonal influx from the stratosphere. The Environmental Measurements Laboratory (EML) in New York routinely measures Be-7 in surface air and their results for 1981 and 1982 from a station at New York, N.Y. (latitude 40°44'N) are plotted in Fig. 28 (EML, 1982 and personal communication). The distribution of Be-7 is in reasonable agreement with the AERU results although levels of Be-7 are higher at New York (average concentrations of 3.1 mBq/m^3 in 1981 and 3.9 mBq/m^3 in 1982). The higher levels at New York can be explained by the slightly larger amount of rainfall which occurs at New York causing increased deposition at this site. Measurements of the

naturally occurring radionuclide, Be-7, provide a quality control check on the air monitoring results and may serve as an indicator of the influence of meteorological phenomena on atmospheric radionuclide levels.

4.2.2.2 Cesium-137

Cesium-137 is a relatively long-lived fission product whose main input to the environment is from nuclear weapons testing that has occurred over the last quarter of a century. Cs-137 and Sr-90 are the two fallout radionuclides that contribute most of the fallout exposure to present generations. Since Sr-90 and Cs-137 have similar half lives, 28.8 yr and 30.2 yr, respectively, and are both produced in similar quantities during nuclear tests with a mean ratio of Cs-137/Sr-90 of 1.5, fallout activity values for one nuclide can be estimated from measurements of the other. Fig. 29 shows the quarterly fallout record for Sr-90 for New York City as measured by EML. The results show that fission product inputs are declining from a peak value in the early 1960's to the relatively low levels present today. The smaller peaks evident in the 1970's are the result of tests carried out by India, France, and the People's Republic of China. Much of the bomb-produced radioactivity is injected initially into the stratosphere where the residence time of particle-reactive radionuclides is in the order of 1 to 2 years.

The Cs-137 results for 1981 and 1982 are plotted in Fig. 28. These results are consistent with low atmospheric levels of Cs-137 in early 1981 with a sharp increase in Cs-137 levels occurring in the spring. This input of Cs-137 resulted from the Chinese weapons test in October 1980. By September 1981, Cs-137 activity had declined to a level below its detection limit and remained so during 1982.

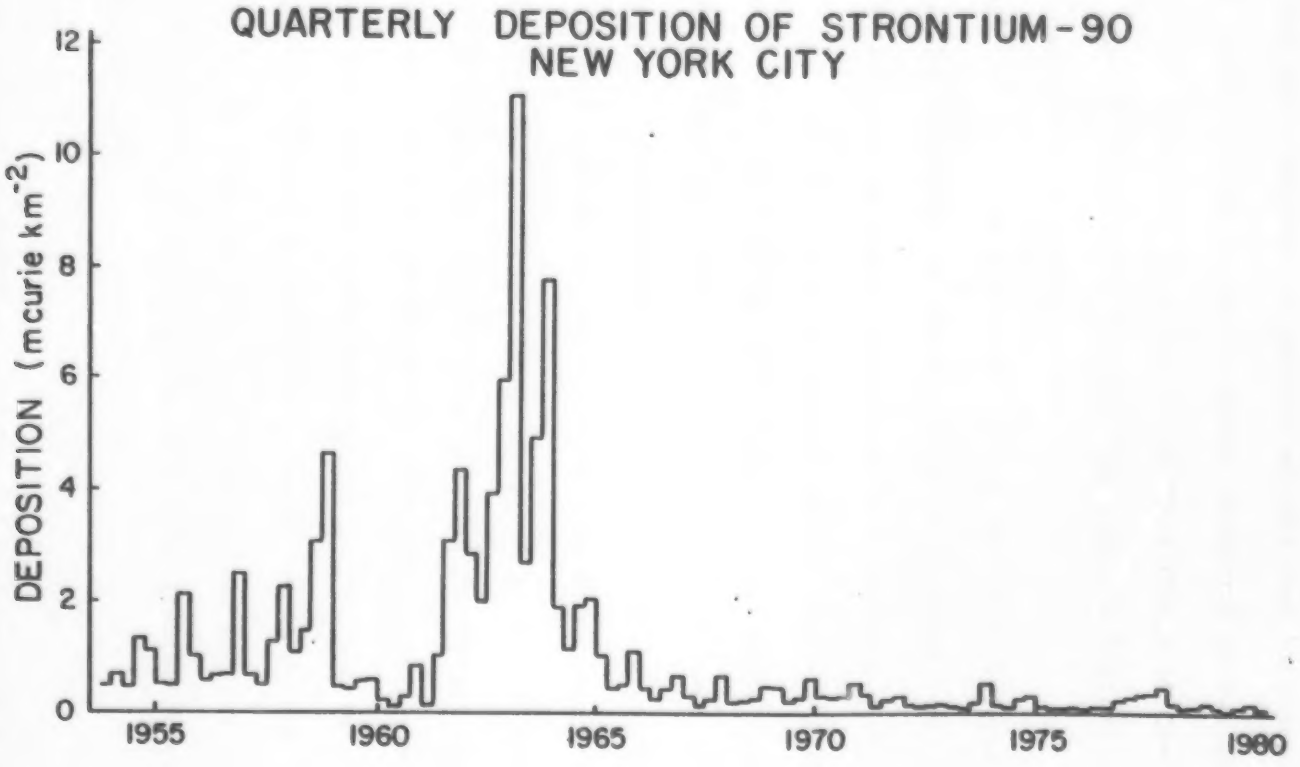


Figure 29. Quarterly fallout record of Sr-90 for New York City measured by the Environmental Measurements Laboratory (1982).

Concentrations of Cs-137 in air, measured in the vicinity of Point Lepreau are compared to Cs-137 air concentrations measured for New York, N.Y. by EML (1982 and personal communication) and again the distribution is similar although levels at New York are higher due to increased atmospheric deposition at this site.

The good agreement between fission products and Be-7 measured in the vicinity of Point Lepreau for 1981 and 1982 and similar data collected by EML at the same time indicates that the air monitoring stations are functioning effectively. Deposition of air particles is estimated to be approximately two-thirds that measured at New York, N.Y. by EML.

5.0 BIOLOGICAL OCEANOGRAPHY

One of the principal concerns associated with the operation of the Point Lepreau NGS is the effect produced on biological phases of the marine environment by the release of radioactivity and waste heat. Radionuclides in the marine environment are subject to a variety of influences. They can remain in solution or in suspension, precipitate and settle to the bottom or be assimilated by plants and animals. Phenomena such as currents, turbulent diffusion, isotopic dilution and biological transport act to dilute and disperse radionuclides while other biological, chemical or physical processes such as the passage of radionuclides from primary trophic levels up through the food chain, physical absorption, ion exchange, flocculation and sedimentation can act as concentrating mechanisms (Rice et al., 1974).

The impact of the thermal plume at the Point Lepreau NGS will mainly be confined to the immediate vicinity of the reactor outfall. The effect may be greatest on the non-mobile benthic community. Eurytherms, such as barnacles and many bivalves can survive large temperature changes as adults, but other stages of growth may be more susceptable to temperature variation. The sublethal effects on organisms may include changes in biological functions such as growth, respiration rate, reproduction and behavior.

The structure of the benthic community around the outfall depends on several factors including salinity, pH, alkalinity and temperature. Any changes in these parameters can signal a change in benthic populations with new species flourishing and non-competitive species dying out. To assess the magnitude of any changes in the benthic community, several studies have been carried out in the Point Lepreau area (described in Smith et al., 1981) including an assessment of the effects of heated water released from

the Coleson Cove thermal generating station located 15 km east of Point Lepreau on the Bay of Fundy coast and an ecological baseline study of benthic communities near the Point Lepreau NGS outfall.

5.1 Biological Uptake of Radioactivity

The availability of radionuclides to marine organisms depends to a large extent upon the physical and chemical states in which they occur in the water. With the exception of tritium, most of the radionuclides released from nuclear reactors are non-conservative elements (i.e. they do not necessarily follow water circulation patterns) and the biological uptake of non-conservative elements is generally higher than that of conservative elements. Marine organisms can accumulate and redistribute radionuclides independently of other physico-chemical processes resulting in the potential "concentration" of radioactivity throughout different levels of the food chain. The term "concentration factor" has been used to describe the biological uptake of chemical species and is defined as the ratio of the concentration of an element in an organism to that concentration directly available from the environment under steady-state conditions. This definition of the concentration factor becomes difficult to apply since most organisms derive their nutrients from a variety of sources too complicated to characterize.

In this report, the concentration factor (C.F.) is defined as the concentration of a radionuclide in the tissue of an organism divided by its concentration in seawater, i.e.;

$$C.F. = (100 - \% H_2O)\ A_{Cs-137\ sample} \,/\, 100\ A_{Cs-137\ water}$$

where; $\% H_2O$ is the percent water per sample wet weight, $A_{Cs-137\ sample}$ is the Cs-137 activity in the sample dry weight and $A_{Cs-137\ water}$ is the

mean Cs-137 activity in sea water in the lower Bay of Fundy (3.5 mBq/l). Concentration factors defined in this way are of limited value for higher organisms in the food chain and for infaunal marine organisms which ingest large amounts of sediment along with organic detritus. In both cases, food sources are varied and inputs of radionuclides can originate from more than one source. Despite these drawbacks, concentration factors do provide a semi-quantitative measurement of the accumulation of radionuclides in different organisms and are useful as indicators of radionuclide release.

The time required for a steady state distribution of a radionuclide to be established between an organism and its environment is related directly to the biological turnover rate of the radionuclide or its stable analogue and is inversely proportional to the rate of environmental dilution. Biological turnover rates are usually expressed in terms of a biological half-life, which is defined as the time required for an organism to exchange half its total inventory of an element. The shorter the biological half life, the faster an organism will reach a steady state with respect to seawater radionuclide concentrations. This concept is important in water masses where radionuclide activities vary as a function of time (such as releases from a nuclear power plant). Organisms high up in the food chain with long biological half lives may never attain a steady state radionuclide concentration with respect to the changing environment. Primary producers such as plankton often have turnover rates on the order of hours, and equilibrium radionuclide levels can change to reflect changing environmental levels.

A third consideration in understanding the degree to which a radionuclide is concentrated by an organism is the extent to which physico-chemical interactions between the element and the organism itself control

the biological uptake process. Goldberg (1957) showed that the ability of marine organisms to concentrate metals from the sea paralleled the order of the strength of metal-organo-ligand complexes, and Szabo (1967) reported that mixed plankton accumulated alkaline earths in the same sequence as cation exchangers such as Dowex-50 in which the order of association is radium > barium > strontium > calcium > magnesium. It is well known that alkali metals (K, Na, Ru, and Cs) may be incorporated into bottom sediments as the result of ion-exchange reactions with clay minerals and it has been proposed that similar ion-exchange reactions may occur at the surfaces of marine organisms. Surface adsorption by ion-exchange may be particularly important for the uptake of alkaline earths and alkali metals by phytoplankton and additional elements may be adsorbed by chelating processes.

5.2 Processing and Analysis of Biological Materials

An assessment of environmental effects associated with the biological uptake of radioactivity requires a surveillance program consistent with local and regional marine energetics, the nature of the food web, water circulation and zones of productivity. Sampling sites have been chosen for the collection of representative species as outlined in Table 15. Biological samples have been collected from the major marine habitats in the Point Lepreau region and these samples have been analysed for radioactivity in the AERU laboratory. Sampling methods, criteria and frequency have been noted in Bishop et al. (1980), although these procedures have been modified to some extent as described below. In general, species which have been selected for surveillance occur abundantly in most of the major habitats and play a significant role in the food web. All trophic levels

Table 15. Representative species for different marine sampling regimes near Point Lepreau.

SPECIES	EMBAYMENT, ESTUARY	INTERTIDAL ZONE	SUBTIDAL ZONE	PELAGIC ZONE
Plants		Rockweed, Irish moss dulse, Fucus	Laminaria	
Grazers	Periwinkle			
Detritus Feeders		Gammarus		
Suspension Feeders	Soft shelled clams	Blue Mussel	Horse Mussel Scallop	
Predators and Scavengers	Sandshrimp	Green crab Hermit Crab	Rock Crab	Rock Crab
Fish		Flounder		Herring
Birds	Purple sandpiper	Eider Duck	Cormorant, Herring Gull	
Mammals			Harbour Seal	
Planktonic Forms				Bulk Samples Phyto- and Zooplankton

are represented and any trends in the transport and accumulation of radioactivity within these communities should become apparent at some level within the sampling assemblage.

Marine biological samples were frozen as soon as possible subsequent to collection and transported in this state to the AERU laboratory. Plant samples were freeze-dried immediately while animal samples were allowed to thaw and specific components such as shell, muscle, liver, brain or hepatopancreas were separated from the tissues. These parts were homogenized (using a food processor or commercial blender) and in some cases, 50 g of material removed for tritium analyses. Soft tissues were freeze-dried while shell and bone material were dried in an oven with the percent moisture content determined for all cases. Subsequent to processing, all samples were kept in air-tight plastic containers.

In order to determine the tritium content of marine biological materials, soils, and terrestrial plants and organisms, it was necessary to extract the water from these samples under non-contaminating conditions. To meet this requirement, an azeotropic distillation procedure (Moghissi et al., 1973) was followed in the AERU laboratory. Briefly, the method involves the addition of scintillation-grade toluene to a solid (or liquid) sample in a distillation flask. With the application of heat, the distillation proceeds at 85°C and the vapour, consisting of 20% water and 80% toluene, is condensed and collected in a vessel where the two components separate. Completion of the extraction is indicated by a rise in the temperature of the vapour to the boiling point of toluene (110°C). The extracted water is then analysed for tritium following the standard liquid scintillation procedure. The remaining bulk sample may be analysed for other radionuclides after removal of the toluene by distillation and

heating in a steam bath. Tritium analyses performed on samples collected prior to 1981 revealed that the biological materials had been contaminated with tritium during handling and storage. Additional samples collected in 1982 were carefully sealed from the atmosphere in plastic bags as soon as possible after sampling to prevent contamination and were used to establish biological baseline levels of tritium.

Samples were analysed for gamma-emitting radionuclides using a 21% efficient germanium-lithium detector. A gamma spectra for each sample was automatically analysed using a PDP-11-04 computer which identified and quantified radionuclide peaks of interest. Samples were analysed using one of five different standard geometries, corresponding to sample volumes of 500 ml, 300 ml, 100 ml, 50 ml, and 20 ml. The effect of self absorption on counting efficiency due to different sample density was found to be negligible for the range encountered in the biological samples. Detection limits (Bq/kg) for each radionuclide of interest are given in Appendices 3 and 4 for the five standard counting geometries employed in these analyses. These detection limits correspond to a counting time of 100,000 sec, for analyses performed one month subsequent to the sampling date (Appendix 3) and one year subsequent to the sampling date (Appendix 4). Note that for the longer time period between sample collection and analysis, many of the shorter-lived radionuclides are virtually undetectable owing to radioactive decay. Appendices 3 and 4 can be used to calculate radionuclide detection limits for any sample listed in the following sections of this report. The standard counting geometry for each sample is used to identify the general detection limit for the isotope of interest from Appendices 3 and 4. This general detection limit is then divided by the sample weight to determine the specific detection limit for the sample.

These detection limits are for typical counting conditions and will vary slightly with changes in the above parameters. The use of these tables as outlined above simplifies reporting procedures because it eliminates the necessity of reporting each radionuclide detection limit for all samples analysed in the AERU laboratory.

5.3 Radionuclide Results - Plankton

Plankton occupy a critical position at the base of the food chain. Their rapid growth rates and large surface areas make plankton an excellent fast-response indicator species for changes in radioactivity levels in sea water. Phytoplankton are primary producers which accumulate radionuclides by both adsorption (predominant mechanism) and absorption. Zooplankton accumulate radionuclides by both processes, as well as by consumption of phytoplankton. In the contaminated waters of the Adriatic Sea (Jelisaveic, 1970) and near Cap de la Hague, France (Germain et al., 1979), plankton have been shown to accumulate Ce-141,144, Ru-106 and Zn-95 at levels up to 3500 Bq/kg. Elevated levels of Cs-137, Co-60, Zn-65, Sb-125, Fe-59 and Mn-54 were also observed. Levels of radionuclides in zooplankton are generally lower than those in phytoplankton owing to their relatively inefficient assimilation of radionuclides across the gut wall (Smith et al., 1981).

Plankton samples were collected during plankton tows along the transects shown in Fig. 30, using plankton nets which were of 64 um mesh size for phytoplankton and 333 um mesh size for zooplankton. Samples were processed as collected, with no attempt to distinguish detritus or suspended particulate matter or to determine the dominant species. The results for gamma-emitting radionuclides in plankton samples are shown in

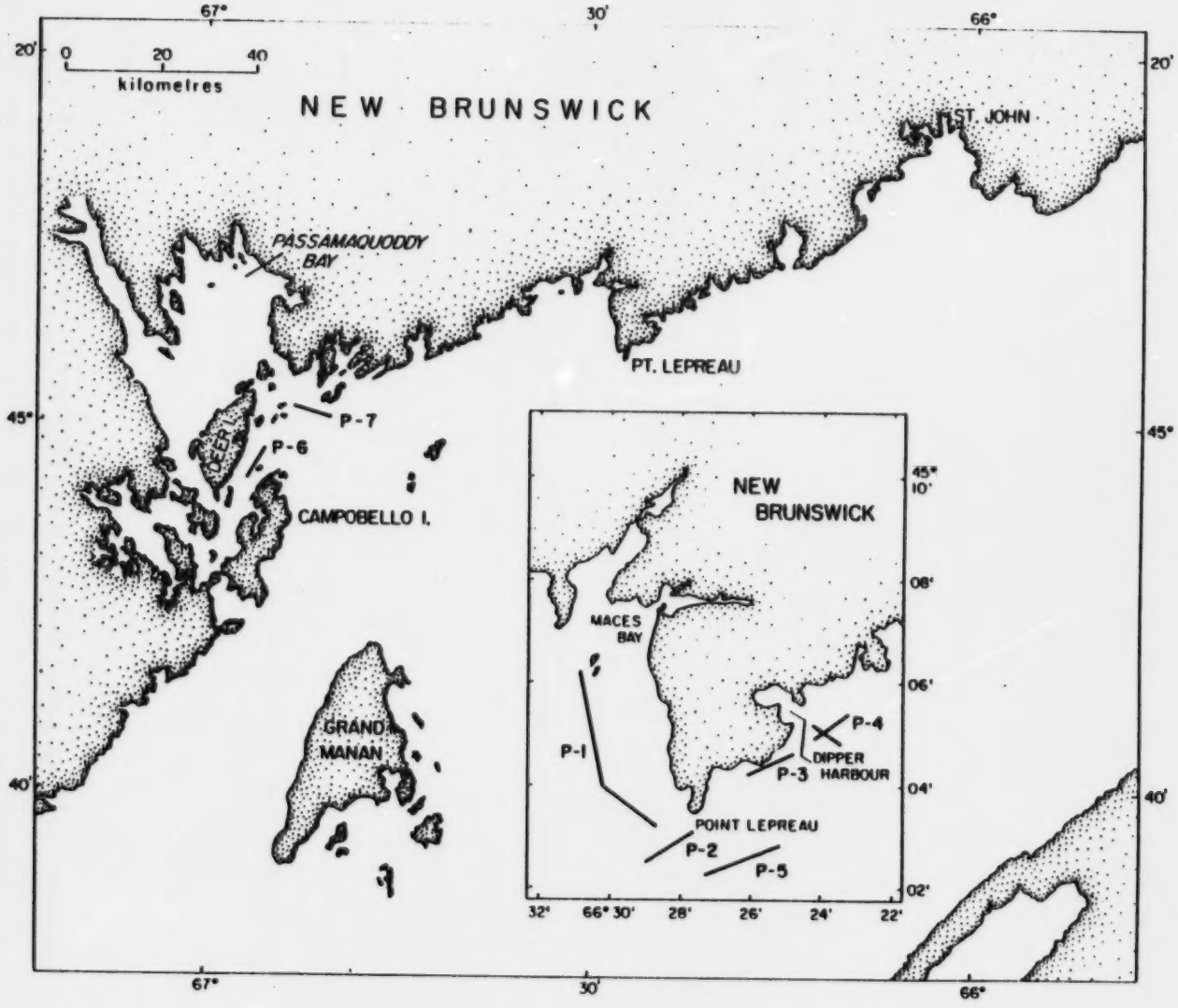


Figure 30. Transects corresponding to sample tows for plankton carried out in the western Bay of Fundy.

Table 16. Artificially-produced radioactivity was measured in only one sample of phytoplankton where Cs-137 was found at a level of 2.3 Bq/kg. The concentration factor calculated for Cs-137 in plankton was in good agreement with literature values.

5.4 Marine Algae

Marine algae play an important role in defining the pathway of radioactive substances in the marine environment. They are primary producers located at the base of a large food web and they have high concentration factors for many non-conservative elements. In addition, large samples are easy to obtain and they are immobile. These factors combine to make them excellent site specific monitors for marine radioactivity and trace elements. The degree of accumulation of radionuclides varies from species to species but most are suitable as indicators.

Marine plants selected as monitoring species at the Point Lepreau NGS include the brown algae; Ascophyllum nodosum, Fucus vesiculosis, and Laminaria Sp. and the red algae; Chondrus crispus and Rhodymenia palmata. In addition, a sample of beach grass and Spartina sp., a chief component of salt marshes, were sampled in 1982.

The radionuclide results for marine plants collected at intertidal (I) and subtidal (S) sites indicated in Fig. 31 are given in Table 17. Cs-137 and Co-60 were measured in several samples while other artificially-produced radionuclides were generally below the detection limit.

It is important to note that in 1981, several algae samples contained short-lived radionuclides Zr-95 and Mn-54. The source of these radionuclides was probably the October 1980 Chinese nuclear weapons test and their detection served to emphasize the value of marine algae as an

Table 16. Radionuclide results for plankton samples. Plankton were not separated from detritus and SPM and the two categories are distinguished solely by the plankton net mesh size.

Organism	Sample No.	Organ	Collection Site	Collection Date	Sample Weight(g)	Sample Geometry	%H_2O	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Co-60 (Bq/kg)
600 µm mesh	80-214	total	P-1	6/2/80						
plankton net	80-215	total	P-2	5/2/80						
sample	80-216	total	P-3	5/2/80	25.76	3	95.8	<2.7	<34	<3.3
('zooplankton')	80-217	total	P-4	8/2/80						
	80-218	total	P-4	8/2/80						
	80-219	total	P-4	8/2/80						
	80-316	total	P-6	24/10/80						
	80-317	total	P-7	24/10/80	72.95	1	97.0	<1.0	<9.0	<1.2
	80-318	total	P-3	3/11/80						
	82-8655	total	P-5	14/5/82	1.67	5	97.0	<12	<121	<15
64 µm mesh	80-220	total	P-1	6/2/80						
plankton net	80-221	total	P-2	5/2/80	18.57	3	96.5	2.3 ± 1.8	<23	<8.0
sample	80-222	total	P-3	5/2/80						
('phytoplankton')	80-223	total	P-4	8/2/80						
	80-224	total	P-4	8/2/80	22.05	3	97.0	<2.0	<18	<3.8
	80-225	total	P-4	8/2/80						
	80-320	total	P-6	24/10/80						
	80-321	total	P-7	24/10/80	29.40	3	96.0	<2.0	<24	<3.5
	80-322	total	P-3	3/10/80						
	82-8651	total	P-5	13/4/82	20.08	3	97.0	<3.1	<31	<3.3
	82-8652	total	P-5	13/4/82	9.05	3	97.0	<7.2	<72	<7.3
	82-8656	total	P-5	13/4/82	81.00	2	95.9	<1.5	<13	<1.8

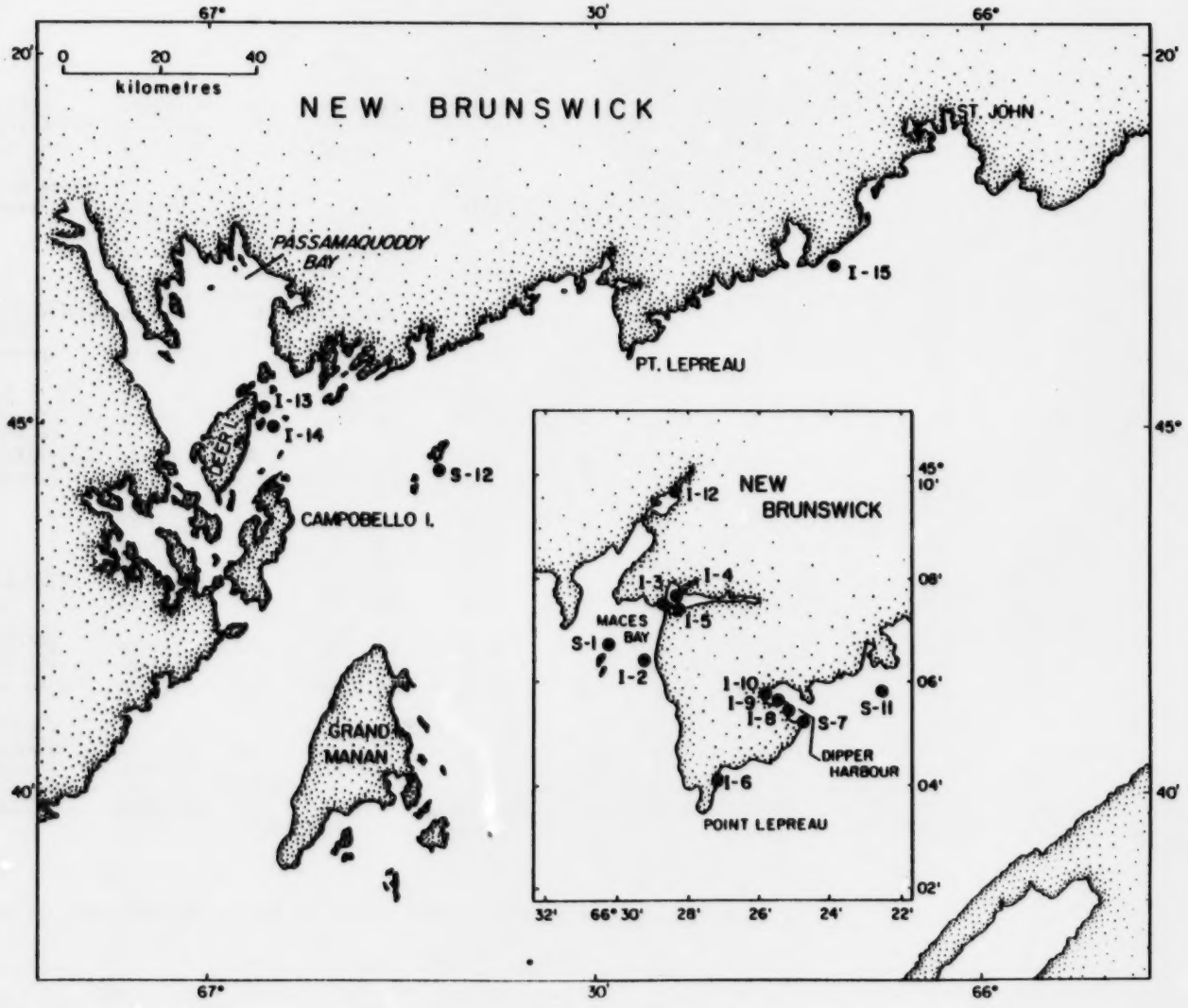


Figure 31. Marine biological sampling sites in the Bay of Fundy intertidal sites (denoted by I) and subtidal sites (denoted by S).

Table 17. Radionuclide results for marine plants collected near Point Lepreau.

Organism	Sample No.	Collection Site	Collection Date	Sample Weight (g)	Geometry	%H_2O	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Co-60 (Bq/kg)	Other Radionuclides (Bq/kg)
Dulse (*Rhodymenia palmata*)	80-233	I-2	7/ 2/80	17.3	3	89.3	<2.7	< 86	<3.2	-
	80-234	I-8	28/ 1/80	25.3	3	81.1	2.2 ± 1.3	123	2.2 ± 1.3	-
Irish Moss (*Chondrus crispus*)	80-235	I-8	28/ 1/80	82.6	2	77.7	<2.0	<134	(1.0	-
	80-236	I-2	7/ 2/80	47.9	3	87.4	<1.0	< 38	<2.0	-
	80-237	I-8	28/ 1/80	127.9	1	76.2	<0.67	< 48	<1.5	-
Kelp (*Laminaria sp.*)	80-238	I-2	7/ 2/80	64.6	1	86.7	<0.67	< 27	<2.0	-
	80-239	I-3	10/ 1/80	24.7	3	87.8	<3.3	<122	<5.0	-
	80-240	I-9	28/ 1/80	62.8	1	92.2	1.2 ± 0.7	27	<0.7	-
	81-681	I-6	1/12/81	59.0	3	84.2	0.5 ± 0.3	24	<0.2	-
	82-9987	I-6	14/10/82	88.32	1	51.4	<0.90	<124	<1.0	-
Rockweek (*Ascophyllum nodosum*)	80-241	I-4	10/ 1/80	133.2	1	76.9	0.4 ± 0.2	29	<0.7	-
	80-242	I-8	28/ 1/80	190.7	1	68.8	<0.5	< 47	<1.3	-
(Growing Tips)	81-680	I-6	1/12/81	129.3	2	76.4	0.3 ± 0.1	21	3 ± 2	Zr-95 2 ± 1
	82-9977	I-6	14/10/82	126.3	1	76.6	0.49 ± 0.38	33	-	-
Rockweed (*Fucus vesiculosus*)	81-682	I-6	1/12/81	157.4	2	76.5	0.18 ± 0.13	13	<0.8	Zr-95 1.2 ± 0.7
	81-683	I-6	1/12/81	43.9	4	74.2	<0.83	< 65	<1.2	Mn-54 0.5 ± 0.3
	82-9980	I-6	14/10/82	93.42	1	76.4	<1.17	< 80	-	-
	82-9979	I-6	14/10/82	96.30	1	84.8	0.34 ± 0.29	10	-	-
	82-9978	I-6	14/10/82	82.22	1	76.4	<1.2	<80	-	-
	82-13389	I-6	14/10/82	95.88	1	76.1	0.40 ± 0.39	27	-	-
Seaweed (Mixed species)	81-684	I-6	1/12/81	31.8	4	69.6	<1.5	<137	<1.8	-
Beach Grass *Spartina sp.* (buds)	82-13374	I-11	20/10/82	42.22	1	85.2	<1.7	<72	<1.9	-
	82-13373	I-11	20/10/82	14.60	3	72.3				
	82-13375	I-11	20/10/82	51.13	1	72.3	1.3 ± 1.1	103	-	-

indicator species. Elevated levels of fallout radioactivity were not detected in 1982 marine plant samples, indicating that their radionuclide inventories had fallen to pre-1981 levels.

5.5 Marine Molluscs

Molluscs can concentrate radionuclides directly from seawater, by ingestion of contaminated food, and by absorption or ingestion of contaminated silt particles or detritus. Molluscs have shown both, an ability to concentrate radionuclides to a marked degree compared to seawater concentrations (Goldberg et al., 1978), and a preference for specific elements (eg. scallops effectively concentrate Mn, Fe is concentrated in the hepatopancreas of oysters) (Pentreath, 1980). In general, molluscs concentrate radionuclides in the hepatopancreas (Schelske, 1973) but other organs such as the kidneys and excretory organs may also play an important role.

The indicator species chosen for the Point Lepreau area were; Mya arenania, a soft shelled clam that has commercial value; Mytilis edulis, or blue muscle; Modiolus modiolus, the horse muscle; Littorina littorea, the common perwinkle; and Pecten maxima (scallops), which is collected as an indicator species for Mn-54.

The results for gamma-emitting radionuclides in molluscs, collected at sites shown in Fig. 31, are given in Table 18. For analysis, mollusc samples were divided into shell and soft tissue. Only Cs-137 and Co-60 were detected and these radionuclides were present at levels very close to the detection limit.

5.6 Marine Crustacea

Radionuclides found in marine crustacea are concentrated mainly in

Table 18. Radionuclide results for marine molluscs collected near Point Lepreau.

Species	Sample No.	Organ	Collection Site	Collection Date	Sample Weight (g)	Sample Geometry	%H2O	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Co-60 (Bq/kg)
Blue Mussel (Mytilis	80-299a	flesh	I-2	6/ 2/80	18.7	3	75.9	<3.3	<241	<4.2
	80-229b	shell	I-2	6/ 2/80	282.8	2	10.5	<0.4	< 90	<0.2
	80-230a	flesh	I-5	10/ 1/80	24.8	3	89.7	<3.2	< 98	2.7 ± 1.7
	80-230b	shell	I-5	10/ 1/80	105.3	3	10.4	<0.6	<179	<0.4
	80-246-53a	flesh	I-15	7-16/ 2/80	17.6	3	75.4	<5.0	<369	<6.7
	80-254-58a	flesh	I-15	7/ 2/80	11.3	3	79.6	<3.7	<224	<5.3
	80-259a	flesh	I-15	7/ 2/80	6.9	4	86.7	<9.8	<392	<12
	80-246-57b	shell	I-15	7-16/ 2/80	351.6	1	14.5	<0.27	< 68	< 0.37
	80-249b	shell	I-15	10/ 2/80	71.1	3	19.5	<1.0	<242	0.5 ± 0.3
	80-250b	shell	I-15	10/ 2/80	74.5	3	17.0	<1.0	<249	<1.5
	80-251b	shell	I-15	16/ 2/80	29.8	3	11.5	<2.7	<708	<3.5
	80-252b	shell	I-15	16/ 2/80	69.1	3	24.4	<1.0	<227	<0.8
	80-258b	shell	I-15	7/ 2/80	56.3	3	9.5	<1.2	<317	1.2 ± 0.7
	80-259b	shell	I-15	7/ 2/80	98.5	3	7.9	<0.8	<230	<1.2
Horse Mussel (Modiolus	80-231a	flesh	S-1	6/ 2/80	58.1	1	88.5	<1.0	< 35	1.0 ± 0.7
modiolus)	80-231b	shell	S-1	6/ 2/80	519.1	1	32.2	<0.07	< 14	0.07 ± 0.05
	80-232a	flesh	I-8	10/ 1/80	42.5	2	83.4	1.7 ± 1.2	83	<1.7
	80-232b	shell	I-8	10/ 1/80	505.5	1	11.8	0.07 ± 0.05	18	0.07 ± 0.05
	81-688	flesh	I-6	1/12/81	21.0	3	82.7	<3.5	<182	<3.8
	81-689	shell	I-6	1/12/81	259.7	2	14.3	0.17 ± 0.15	43	0.32 ± 0.20
	82-13387	shell	I-6	14/10/82	485.6	1	16.8	<0.11	< 30	<1.3
	82-13388	flesh	I-6	14/10/82	34.7	3	53.5	<1.28	<180	<1.5
Soft Shelled Clam	80-226a	flesh	I-5	10/ 1/80	82.3	1	87.6	0.5 ± 0.2	25	<0.7
	80-226a	shell	I-5	10/ 1/80	193.9	1	11.1	<0.3	<133	<0.7
(Mya arenaria)	80-227a	flesh	I-10	6/ 2/80	24.2	3	83.5	2.5 ± 1.7	124	<2.2
	80-227b	shell	I-10	6/ 2/80	312.5	1	26.0	0.7 ± 0.2	148	<0.5
Periwinkle	81-686	flesh	I-6	1/12/81	9.7	3	73.4	<4.5	<359	2.8 ± 2.5
(Littorina	81-687	shell	I-6	1/12/81	358.5	2	3.6	<0.3	< 96	<0.2
littorea)	82-9985	shell	I-6	14/10/82	477.64	1	3.8	<0.03	< 13	-
	82-9986	flesh	I-6	14/10/82	87.16	2	77.5	<0.71	< 46	<0.78
Scallop	82-13382	muscle	I-6	14/10/82	41.95	3	52.6	<1.23	<176	<1.6
(Placopecten	82-13383	gut	I-6	14/10/82	38.61	3	59.5	<1.89	<216	<2.2
magellanicus	82-13381	shell	I-6	14/10/84	514.23	1	17.4	<0.11	< 30	<0.11

the heptopancreas, gills, digestive glands and in the shell (Van Weers, 1975; Pentreath, 1980).

The crustacea selected for use in the Point Lepreau monitoring program include Carcinus maenas, the green crab; Cancer sp., rock crab; Pagurus sp., hermit crab; Crangon sept mspinosus, sandshrimp; and Gammarus oceanicus, amphipods.

The radionuclide results for crustacea are shown in Table 19. Cs-137 was detected in sandshrimp, rock crab flesh and green crab hepatopancreas. From the results in Table 19, sandshrimp appears to be the most suitable monitoring species as both Cs-137 and Co-60 were easily detectable.

5.7 Marine Vertebrates

In general, marine vertebrates are less suitable as indicator species because of their mobility and their inefficient accumulation of radionuclides compared to other organisms. On the other hand, they are commercially important and occupy a vital position relative to man in the food web. Marine vertebrates accumulate radionuclides by both absorption from food or other material passing through the gut and direct adsorption from solution.

Marine vertebrates selected for monitoring near Point Lepreau include the following: harbour seal, Phoca vitulina; harbour porpoise, Phocoena phocoena, when available; Pleudopleuronectes americanus, the winter flounder; and Clupea harengus, the atlantic herring. In general levels of radionuclides in marine vertebrates were low (Table 20) and the results indicate that there is no apparent increase in activity with increasing trophic level of the organism (see Table 15). The low

Table 19. Radionuclide results for marine crustacea and Echinoderms collected near Point Lepreau.

Organism	Sample No.	Organ	Collection Site	Collection Date	Sample Weight	Sample Geometry	% Water	Cs-137 (Bq/kg)	Concentration Factor	Co-60 (Bq/kg)
Crustacea										
Sandshrimp (*Crangon sp.*)	80-312	total	I-3	17/9/80	56.1	2	59.6	0.8 ± 0.5	101	0.8 ± 0.5
Rock Crab (*Cancer sp.*)	80-314a	flesh	I-6	16/9/80	23.4	3	74.4	<1.2	<90	1.0 ± 0.7
	80-314b	shell	I-6	16/9/80	245.6	1	54.9	<0.3	<45	<0.5
	80-314c	hepato-pancreas	I-6	16/9/80	8.6	5	67.4	<2.8	<277	<4.0
Green Crab (*Carcinus sp.*)	81-685a	flesh	I-6	1/12/81	8.7	5	78.3	<6.8	<326	<8.2
	81-685b	shell	I-6	1/12/81	57.4	3	59.0	<5.2	<451	<8.3
	81-685c	hepato-pancreas	I-6	1/12/81	10.5	4	77.9	<9.8	<652	<6.8
	80-313a	flesh	I-8	7/9/80	10.9	4	60.0	<5.0	<600	<8.3
	80-313b	shell	I-8	7/9/80	168.3	1	61.5	<0.8	<96	<0.8
	80-313c	hepato-pancreas	I-8	7/9/80	7.2	4	71.3	<10.0	<860	6.7 ± 5.0
	82-13384	total	I-8	14/10/82	71.3	2	71.5	<1.5	<120	<1.5
Hermit Crab (*Pagurus sp.*)	80-228a	flesh	S-7	31/1/80	14.7	4	73.5	1.5 ± 1.2	120	<4.0
	80-228b	shell	S-7	31/1/80	34.1	3	61.6	1.7 ± 1.0	192	<2.0
Amphipod (mixed)	81-690	total	I-6	1/12/81	8.7	4	81.0	<8	<456	<4.2
Echinoderms										
Sea Urchin (*Stongylocentrotus droebachiensis*)	82-13380	total	I-6	14/10/82	247.2	1	69.5	<0.24	<20	<3

Table 20. Radionuclide results for marine vertebrates collected near Point Lepreau.

Species	Sample No.	Organ	Collection Site	Collection Date	Sample Weight	Sample Geometry	%H2O	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Co-60 (Bq/kg)
Harbour Seal (Phoca vitulina)	80-310	muscle	I-13	16/7/80	21.9	3	69.8	1.7 ± 1.5	151	<2.0
	80-327	muscle	I-13	16/7/80	34.1	3	56.1	<1.3	<176	<1.6
	80-243	liver	I-13	16/7/80	155.4	1	67.8	2.0 ± 0.3	193	1.0 ± 0.2
	80-311	liver	I-13	16/7/80	127.8	2	70.5	1.5 ± 0.3	133	<0.5
	80-328	liver	I-13	16/7/80	97.7	2	60.9	1.2 ± 0.5	137	<1.2
	80-245	brain	I-13	16/7/80	41.3	2	78.7	2.3 ± 1.3	149	<3.7
	80-308	brain	I-13	16/7/80	22.0	3	77.0	<2.2	<150	3 ± 1
	80-244	adipose (fat)	I-13	16/7/80	107.2	2	10.9	<1.2	<312	<1.5
	80-309	adipose	I-13	16/7/80	201.5	2	54.3	0.3 ± 0.2	46	<2.0
	80-326	adipose	I-13	16/7/80	236.3	2	32.4	0.3 ± 0.2	68	<2.0
	80-307	throat	I-13	16/7/80	6.2	4	69.3	<4.3	<399	9 ± 5
	80-325	gland								
Herring (Clupea harengus)	80-323a	muscle	S-12	30/12/80	99.6	1	69.7	1.2 ± 0.5	106	<1.8
	80-323a	muscle	S-12	30/10/80	100.3	2	69.3	1.0 ± 0.7	92	<1.8
	80-323c	kidney+ liver	S-12	30/10/80	3.7	5	61.0	<7.5	<697	<11
	80-323d	head	S-12	30/10/80	8.8	4	68.5	<7.0	<662	8 ± 4
	80-323e	gill	S-12	30/10/80	7.0	4	72.1	<9.7	<809	<12
	80-323f	bone	S-12	30/10/80	6.3	4	63.3	<11	<1200	8 ± 5
	80-323g	gut	S-12	30/10/83	24.2	4	54.5	<3.0	<410	<4.0
Winter Flounder (Pseudopleuronectes americanus)	80-315	muscle	S-11	30/10/80	45.2	2	72.9	<1.5	<122	<12
	82-09982	flesh	I-6	14/10/82	118.08	1	79.4	1.0 ± 0.2	60	-
	82-09983	gut + bone	I-6	14/10/82	136.18	1	76.6	0.4 ± 0.1	30	-
	82-09984	skin	I-6	14/10/82	39.59	2	75.6	<1.9	<132	<2.1

radioactivity levels measured in marine organisms are due to the enormous diluting effect of the sea water reservoir.

5.8 Tritium in Marine Organisms

Most of the tritium released from nuclear reactors is in two forms: molecular tritium gas, released to the atmosphere via stack gas discharges and triliated water (HTO) both in vapor and liquid form. Tritium released in the form of molecular tritium is converted into HTO by either exchange or oxidative reactions. The HTO interacts with living organisms via two pathways. The first and most direct is exposure of organisms to water contaminated with HTO and more indirectly the ingestion by organisms of foods contaminated with tritium.

Tritium is incorporated by marine and aquatic organisms as tissue-free water and tissue-bound water (i.e. incorporated into organic molecules) and incorporation is dependent on the physico-chemical state of the tritium in the liquid effluent. Organisms exchange tritiated water, HTO, with free tissue water very quickly. This short-lived component with a half-life on the order of days (IAEA, 1981) can represent up to 95% of the tritium in the organism. On the other hand, tritium can be incorporated into organic compounds by the exchange reaction with H in organic compounds. This organically-bound tritium fraction consists of two components, a rapidly exchanged fraction with a biological half-life on the order of hours or days and a slowly exchangeable fraction with a $t_{1/2}$ on the order of 300 days (IAEA, 1981). Studies on natural ecosystems revealed levels of tritium incorporated into organic matter 20 times higher than levels in the surrounding water. These seemingly anamolously high levels were due to the presence of tritiated organic compounds in the effluent

that were readily absorbed and only slowly exchanged with tritium in the environment. Thus it would appear that the physico-chemical state of the tritium is a sensitive parameter of particular importance for predicting and evaluating the environmental behavior of tritium.

The effects of tritium incorporation on plants and animals results from mass difference, beta particle emission or the properties of the decay product He-3, which differ vastly from the properties of hydrogen. Tritium in vivo is exchangeable with hydrogen and can replace hydrogen in any biochemical compound. It is also possible for tritium to be incorporated into non-exchangeable positions in biological molecules including proteins, DNA and RNA. Funk and Persen (1972) have shown that the decay of tritium in cytosine resulted in specific mutation and Vig and McFarlane (1972) have shown genetic effects in soybean plants when the seeds were germinated in water containing as little as 370 Bq/ml of tritium. In general, the concentrations of tritium found in the environment are such a small fraction of the total water content that the probability of observing any radiation effects in biological systems are low.

Tritium levels were determined in a variety of marine organisms collected in the vicinity of Point Lepreau using the azeotropic distillation method outlined in Section 5.2. (Table 21). Contamination problems encountered previously in sampling and analytical procedures have been eliminated. Tritium levels measured in marine plant and animal samples with the exception of shell material of molluses and crustacea were all within the range of tritium activities in seawater (<1.5 Bq/l to 3.4 Bq/l). This is in agreement with work by Kirchmann (1979) who showed that levels of body-water tritium in the marine invertebrate Katelysia opima reached maximum concentration factors of 0.88. The highest tritium levels

Table 21. Tritium levels found in Marine Systems.

Species		Sample No.	Site No.	H-3 (Bq/l)	Concentration Factor
Water		-	-	avg 2.31 Bq/l*	
Algae					
Fucus Vesiculus		82-09979	I-6	3.1 ± 1.6	1.3
		82-09978	I-6	<3.02	
Laminaria		82-09977	I-6	2.3 ± 1.8	1.0
		82-13389	I-6	<3.02	-
Ascophyllum		82-09981		3.2 ± 1.6	1.4
Salt Marsh					
Marsh Grass		82-13374	I-11	3.0 ± 1.8	1.2
Spartina		82-13373	I-11	<3.02	-
Mollusc					
Pecten maximum	Shell	82-13381	I-6	5.4 ± 1.8	2.3
	Muscle	82-13382	I-6	2.1 ± 1.7	0.9
	Gut	82-13383	I-6	2.2 ± 1.7	1.0
Modiolus	Shell	82-13387	I-6	4.3 ± 1.7	1.9
Modiolus	Gut	82-13388	I-6	>3.02	-
Crustacea					
Carcinus sp.	Muscle & Gut	82-13384	I-6	2.4 ± 1.7	1.0
	Shell	82-13385	I-6	6.3 ± 1.8	2.7
Marine Vertebrates					
Pseudopleuronectes americanus	Flesh	82-09982	I-6	2.7 ± 1.6	1.2

* Range of tritium values found in seawater: <1.5 Bq/l to 3.4 ± 1.8 Bq/l

were found in shell material of molluses and crustacea. Concentration factors of 1.9 to 2.7 were observed in shell material compared to concentration factors of 0.9 to 1.4 for the remainder of the samples. These high concentration factors for shell material have not been previously reported in the literature and further analysis of shell material will be undertaken to verify this observation.

Tritium levels observed for the complete sample set gave no indication that tritium is concentrated in higher levels of the marine food chain.

6.0 TERRESTRIAL AND AQUATIC ENVIRONMENTAL PHASES

The complex interrelationship between the terrestrial and aquatic enviromental phases is illustrated in Fig. 32. Radionuclides are injected into the system via two primary pathways: (1) the atmosphere, from stack discharges at nuclear generating stations and from above ground nuclear tests and (2) from effluent releases in the cooling water outflow. In both cases, molecular and turbulent diffusion processes take effect immediately and generally act to reduce radionuclide concentration as a function of distance from the source.

At the same time the radionuclides are subject to depositional phenomena, including gravitational settling, precipitation scavenging, impaction, chemical adsorption and exchange. The high particle reactivity of some radionuclides leads to their rapid removal from the air-water continuum and plays an important role in their subsequent availability for biological uptake.

Radionuclides in the abiotic phases of the environment can enter the food chain at any trophic level. Plants, which are at the base of the grazing food chain, accumulate radionuclides from the soil, atmospheric deposition and sorption from water. At higher trophic levels, mechanisms such as the inhalation of aerosol particles can play an important role for some insoluble radionuclides, as well as the standard plant-herbivore-carnivore cycles.

Ultimately, radionuclides tend to accumulate in "sinks" composed of deeper strata of soils, sediments and their associated deposits of organic detritus where radionuclides are tightly bound and have negligible contact with biological components. The environmental distribution of radioactivity is as much a function of physiological partioning within the biotic

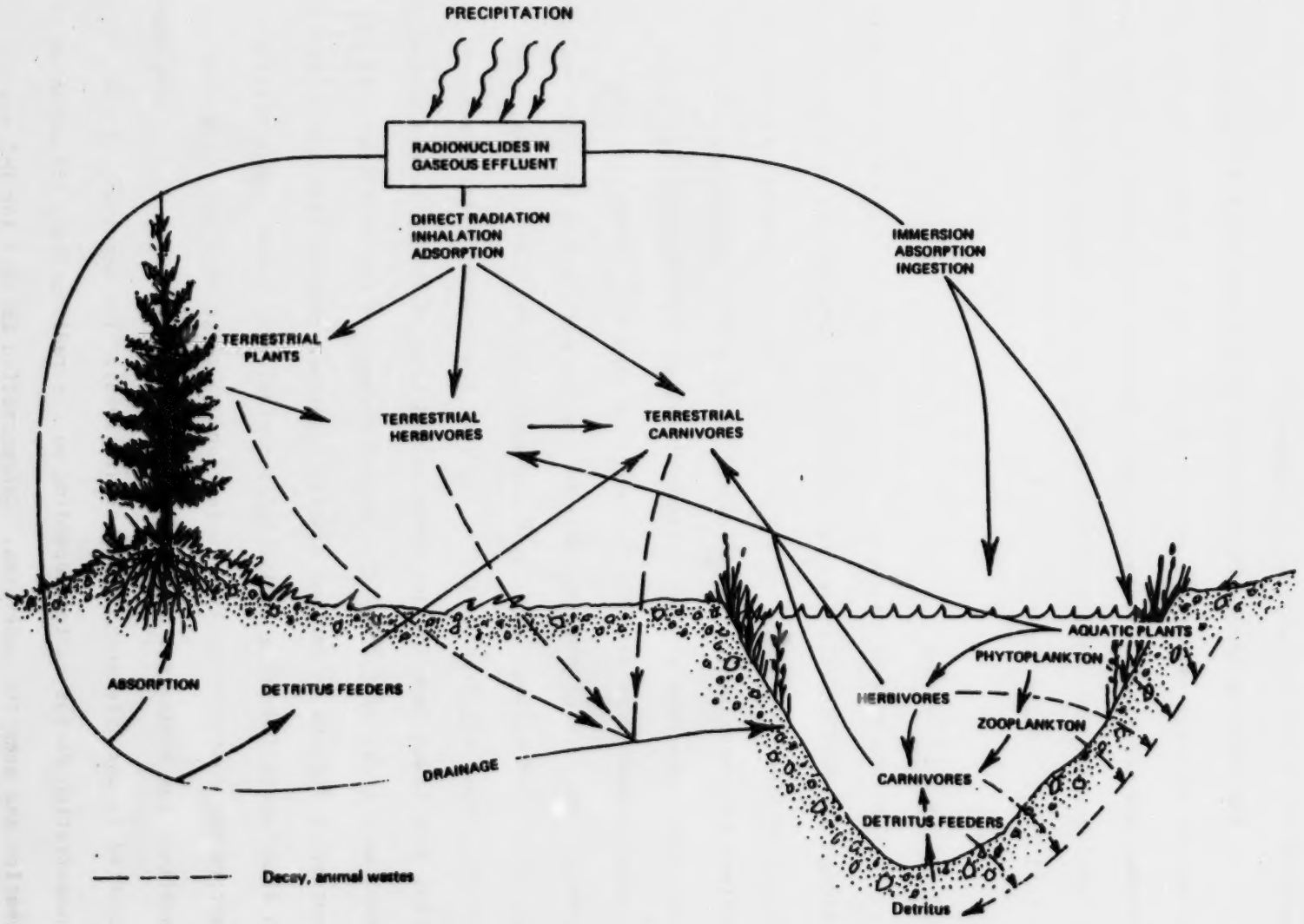


Figure 32. Illustration of transport pathways for radionuclides in the terrestrial and aquatic environments.

compartments as it is a function of the physical and chemical interactions between the biotic and abiotic compartments.

The rates at which the radionuclides are distributed throughout an ecosystem determine the speed with which their steady state distribution is reached and the final distribution patterns that are obtained. The factors governing these processes will be discussed in the context of the baseline experimental results for Point Lepreau.

6.1 Aquatic Systems

The transfer of radionuclides through aquatic systems is controlled through the complex relationship of physical, chemical and biological processes inherent in an aquatic habitat (Fig. 32). In general, aquatic systems are characterized by a central aqueous reservoir in contact with the other components of the environment. This type of structure makes aquatic systems more suitable for modeling than most ecosystems. In general, the fate of radionuclides in aquatic systems has been studied using long-lived biologically-important radionuclides such as Sr-90 and Cs-137. Tracy and Prantl (1982), studying the fate of fission product input into Lakes Superior and Huron, found that, although bottom sediments acted as a sink, Cs-137 was distributed throughout the ecosystem with concentration factors in various aquatic organisms ranging from 400 to 6000. In experiments involving artificially contaminated lakes, concentration factors for most radionuclides were found to range over many orders of magnitude for biological materials and values from 100 to 25,000 have been reported in the literature (Polikarpov, 1966). The magnitude of the concentration factor varies depending on the radionuclide, the organism in question and even the ecosystem. Concentration factors for H-3 are

generally less than one for most organisms (IAEA, 1981) while concentration factors of up to 300,000 have been measured for P-32 in algae (Kornberg and Davis, 1966). In general, lakes which are oligotrophic (low nutrient levels, comparatively high dissolved oxygen) are inhabited by organisms having higher radionuclide levels than those found in eutrophic lakes (high nutrient levels, low or zero dissolved oxygen).

The five sampling sites, T-1 - T-5, selected for collecting aquatic samples are identified in Fig. 33. Two of the sites (T-2, T-4) are small lakes and the remaining three sites are stream locations. The experimental methodology for the preparation of biological, sediment and water samples and subsequent analysis using nuclear instrumentation are similar to procedures reported earlier in this report (Section 5.2). The principal difference is that the analytical problems associated with the salt content of marine samples are not encountered in the present case.

6.1.1 Water Column Results

Water samples were taken at the five aquatic sampling sites indicated in Fig. 33. Samples were analyzed for both H-3, which exhibits almost no partitioning between dissolved and particulate phases in aquatic systems, and Cs-137, where up to 90% of which can be found in the sediments.

Cs-137 tends to follow particle pathways through aquatic systems with its overall distribution depending on the size and physico-chemical characteristics of the aquatic system (Smith and Ellis, 1982). Cs-137 levels in the water ranged from <1.2 mBq/l to 5.5 mBq/l (Table 22) with no clear distinction between stream and lake concentrations. This is to be expected as the water shed is similar throughout the Point Lepreau region.

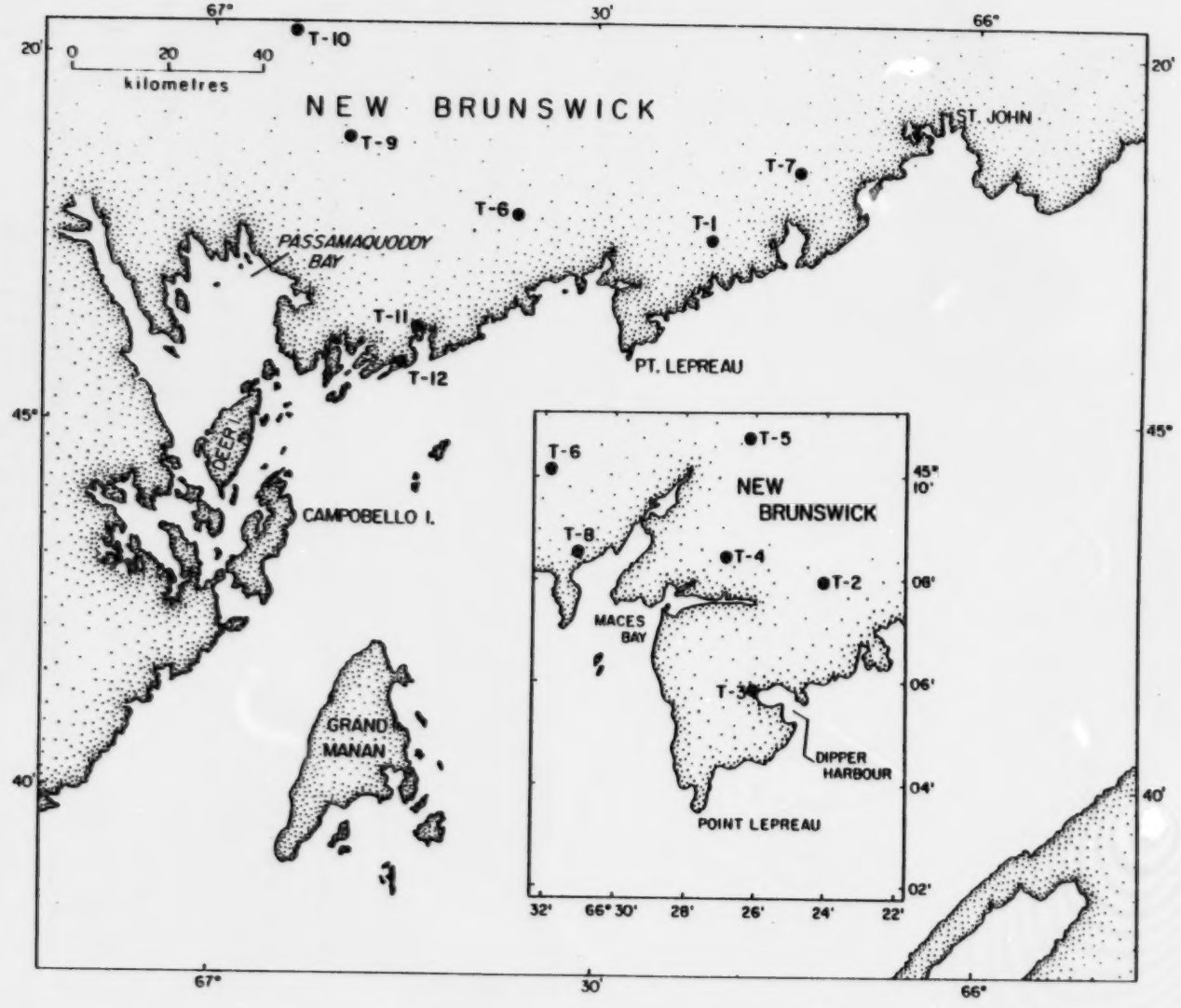


Figure 33. Terrestrial and aquatic sampling sites in vicinity of Point Lepreau.

Table 22. Cesium-137 and H-3 results for water and suspended particulate material from aquatic sites near Point Lepreau, 1981 and 1982.

Sample No.	Sampling Site	Suspended Matter Level (mg/l)	H-3 (Bq/l)	Cs-137 Solution (mBq/l)	Particles (mBq/l)	Particles (Bq/g)	Total Cs-137 (mBq/l)	Distribution Coefficient (K_d) (Bq/g/Bq/ml)
81-04642	T-2	0.97	4.2 ± 1.2	1.7 ± 0.3	2.8 ± 1.3	2.9 ± 1.6	4.5 ± 2.3	1.7×10^6
81-04643	T-2	0.74	-	2.3 ± 0.5	3.2 ± 1.5	4.1 ± 2.1	5.5 ± 2.8	1.8×10^6
81-04644	T-2	1.53	-	2.0 ± 1.0	2.2 ± 1.0	1.6 ± 0.7	4.2 ± 2.4	0.73×10^6
81-04645	T-3	0.61	6.0 ± 1.2	>0.8	0.4 ± 0.6	0.7 ± 1.0	>1.2	0.8×10^6
81-04646	T-4	1.70	5.3 ± 1.2	>0.8	0.8 ± 1.4	0.5 ± 0.8	>1.6	0.6×10^6
81-04647	T-5	1.04	4.5 ± 1.2	3.3 ± 0.5	1.2 ± 1.0	1.3 ± 0.9	4.5 ± 3.8	0.34×10^6
81-04648	T-1	1.00	6.1 ± 1.3	1.0 ± 0.7	0.5 ± 0.7	0.5 ± 0.7	1.5 ± 2.2	0.53×10^6
82-13367	T-2	-	10.6 ± 2.3	-	-	-		-
82-13376	T-1	-	9.8 ± 1.8	-	-	-		-
82-13377	T-3	-	6.1 ± 1.8	-	-	-		-
82-13378	T-4	-	6.2 ± 1.8	-	-	-		-
82-13379	T-5	-	5.0 ± 1.8	-	-	-		-

An average of 46% of the Cs-137 was found on particles (range of 27-62%).

Tritium levels were very uniform for the five sites studied, with activities in the range of 4.2 Bq/l to 6.1 Bq/l and a mean tritium activity of 5.2 ± 1.2 Bq/l for 1981 and in the range of 5.0 Bq/l to 10.6 Bq/l with a mean tritium activity of 7.5 Bq/l in 1982. This is in good agreement with the value of 5.9 ± 3.8 Bq/l found in water vapor in the 1981 air monitoring station results and with the value of 5.8 ± 2.4 Bq/l found in 1982. This agreement is indicative of the lack of partioning of H-3 subsequent to its introduction into the continential hydrological cycle.

6.1.2 Sediment Results

Retreat lake is a small privately owned lake, 9 km northeast of Point Lepreau. It is situated in a likely position to intercept any airborne radionuclides transported by the predominant southwesterly winds, which are most prominent in the summer (see Table 11 and 12). The location and ease of acceqs make it a good sampling location although the shallowness and the recreational use of the lake may cause some degree of mixing in the sediment. In 1981, eight sediment cores were taken and subsequently analyzed at AERU laboratory. A detailed discussion of the sediment results is presented in a previous report (Smith et al., 1982). In summary, the results of the core analysis indicate that particle-reactive radionuclides released from the Point Lepreau NGS may be efficiently retained in Retreat Lake. The inventories of these radionuclides in the sediment may prove useful as a guide to determining the total fluxes of radionuclides from the Point Lepreau NGS. Unfortunately, the lake sediments are subject to active mixing phenomena which disturb the geochronology of the system. Future plans include sediment sampling of a second lake, Clear Lake, a deeper,

more isolated lake which is located approximately 3 km west of Retreat Lake and may provide a more well-preserved depositional record.

6.1.3 Aquatic Plants and Animals

Radionuclide results for aquatic plants and animals collected near Point Lepreau are given in Tables 23 and 24. Concentration factors (C.F.) for these samples have been estimated for samples containing Cs-137 using the mean Cs-137 activity measured in fresh water systems near Point Lepreau ($\bar{x}$ = 2.7 mBq/l, Table 22). These concentration factors are a useful means of making qualitative comparisons of radionuclide uptake by different organisms in the same aquatic system. Detection limits can be estimated using Appendices 3 and 4.

Aquatic plants and invertebrate animals accumulate radionuclides by adsorption and absorption from air and water. Vertebrates, on the other hand, accumulate radionuclides mainly by ingestion and are much slower to respond to changes in radionuclide concentrations in the environment.

Levels of artifical radionuclides measured in aquatic plants in 1982 are generally lower than those measured in 1981 (Table 23). For example, the average Cs-137 activity in wild iris in 1982 was 7.2 Bq/kg while in 1981 the average activity was 33 Bq/kg. The short-lived Zr-95, which was produced during the October 16, 1980 Chinese nuclear test and was measured in 1981 samples, was not detected in 1982 samples. Ce-144, present at an average level of 57 Bq/kg in horsetail and of 30 Bq/kg in iris in 1981, was measured in only one sample of horsetail in 1982, at a level of 24 Bq/kg. The decrease in activity of fallout radionuclides including the long-lived nuclide Cs-137 over the period of a year is a result of the growth patterns of these plants in addition to radioactive

Table 23. Radionuclide results for aquatic plants collected near Point Lepreau.

Species	Sample No.	Component	Collection Site	Collection Date	Sample Weight	Sample Geometry	%H$_2$O	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Zr-95 (Bq/kg)	Ce-144 (Bq/kg)	Be-7 (Bq/kg)	Mn-54 (Bq/kg)
Horse Tail (*Equisetum sp.*)	81-4673	leaves	T-1	14/7/81	39.8	1	76.9	113 ± 7	1.3×10^4	16 ± 8	25 ± 15	113 ± 53	4.7 ± 2.2
	81-4674	leaves	T-2	13/7/81	42.9	1	86.5	230 ± 8	1.6×10^4	10 ± 5	43 ± 17	93 ± 50	<4.2
	81-4675	leaves	T-3	13/7/81	29.1	2	77.6	5 ± 2	560	15 ± 10	48 ± 18	77 ± 43	<5.5
	81-4676	leaves	T-4	13/7/81	28.7	2	74.8	16 ± 5	2.0×10^3	15 ± 10	58 ± 17	93 ± 50	<5.5
	81-4677	leaves	T-5	14/7/81	44.7	2	79.6	8 ± 3	820	<20	48 ± 15	165 ± 47	<5.2
	82-13354	total	T-1	7/7/82	31.3	1	82.1	63 ± 5	5.6×10^3	<12	24 ± 3	<72	<4
	82-13355	total	T-2	7/7/82	27.3	1	80.5	6 ± 4	585	<13	<31	90 ± 63	<4
	82-13356	total	T-3	7/7/82	24.1	1	79.8	5 ± 4	505	<17	<37	<91	<5
	82-13357	total	T-4	7/7/82	36.6	1	76.0	<4	480	<230	<47	<2500	<6
	82-13358	total	T-5	7/7/82	42.5	1	75.4	<2	240	<772	<36	-	<5
Wild Iris (*Iris versicolor*)	81-4669	total	T-2	14/7/81	31.9	1	82.2	55 ± 7	4.9×10^3	25 ± 25	36 ± 25		
	81-4671a	leaves	T-4	13/7/81	47.4	1	70.9	16 ± 4	2.3×10^3	<22	25 ± 12	11 ± 7	
	81-4671b	roots	T-4	13/7/81	16.1	4		28 ± 8	4.1×10^3	<43	<43	<96	
	82-13349	total	T-1	7/7/82	49.1	1	80.2	18 ± 2	1.8×10^3	< 5	<14	27 ± 29	<2
	82-13350	total	T-2	7/7/82	34.1	1	69.0	8 ± 3	1.2×10^3	<10	<25	<59	<4
	82-13351	total	T-3	7/7/82	40.5	1	81.9	3 ± 3	272	<11	<23	141 ± 56	<3
	82-13352	total	T-4	7/7/82	48.5	1	81.7	4 ± 2	366	< 9	<54	<54	<3
	82-13353	total	T-5	7/7/82	45.6	1	83.9	3 ± 1	242	-	-	-	-
Water Lily (*Nuphar sp.*)	81-4668	total	T-1	14/7/81	61.1	1	82.0	39 ± 3	3.5×10^3	20 ± 10	<37	-	1.8 ± 1.5
	81-4672a	leaves	T-5	14/7/81	38.2	1	93.3	79 ± 4	2.7×10^3	<26	<37	-	2.3 ± 1.7
	81-4672b	roots	T-5	14/7/81	9.7	4		30 ± 5	1.0×10^3	<74	<75	-	<12
	81-4691a	leaves	T-5	14/7/81	44.5	1		85 ± 6	1.7×10^3	<33	<130	-	<5.5
	81-4691b	roots	T-5	14/7/81	25.8	1	96.1	28 ± 6	550	<45	<5.3	-	<7.1
Cattail (*Typha sp.*)	81-4670	total	T-3	13/7/81	23.6	1	88.0	<13	<80	<41	<57	-	<6.8
Pond weed (*Pontamogeton*)	81-4678a	leaves	T-1	14/7/81	5.3	4	43.1	<13	$<4 \times 10^3$	<200	<120	-	<20
	81-4678b	roots	T-1	14/7/81	42.9	2	55.1	32 ± 3	7.2×10^3	<53	<49	-	<5.5
	81-4688	total	T-4	13/7/81	10.4	4	62.6	23 ± 3	4.3×10^3	66 ± 25	23 ± 11	-	35 ± 22
Lake vegetation (sediment surface)	81-4649	total	T-2	22/7/81	42.1	1	70.2	28 ± 5	4.2×10^3	<13	<37	-	<4.6

Table 24. Radionuclide results for aquatic organisms collected near Point Lepreau.

Organism	Sample No.	Organ	Collection Site	Collection Date	Sample Weight (g)	Sample Geometry	%H_2O	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Co-60 (Bq/kg)
Frog (*Rana clamitans*)	81-4679	total	T-1	14/7/81						
(*Rana sylvatica*)	81-4685	total	T-3	13/7/81	6.53	4	80.0	32.0 ± 6	3.2×10^3	13
(*Rana clamitans*)	81-4687	total	T-4	13/7/81						
(*Rana catesbyiana*)	81-4692	total	T-5	14/7/81	7.48	4	85.0	49.0 ± 9	3.7×10^3	12
(*Rana catesbyiana*)	81-4684	total	T-2	13/7/81	13.84	4	78.9	24.0 ± 3	2.5×10^3	4.0
Fish (*Cyprinidae* +	81-4682	total	T-2	13/7/81	4.3	5	88.0	12.0 ± 5	720	7.0
Pungitius pungitius)	81-4690	total	T-5	14/7/81						
Cyprinidae	81-4689	total	T-4	13/7/81	5.3	4	86.8	5.5 ± 4.0	360	17
Cyprinidae	81-4680	total	T-1	14/7/81	17.6	3	87.5	17.0 ± 4	1.1×10^3	4.7
Insects (peepers)	81-4686	total	T-4	13/7/81	0.20	5	5.5	143	1.4×10^3	100
	82-13316	total	T-1	7/7/82	2.86	5	64.79	8	1.4×10^3	-
	82-13317	total	T-2	7/7/82	14.52	3	74.09	3	388	-
	82-13318	total	T-3	7/7/82	2.06	5	68.15	12	1.9×10^3	-
	82-13319	total	T-4	7/7/82	10.03	3	62.93	14.0 ± 4	2.6×10^3	-
	82-13320	total	T-5	7/7/82	28.19	2	55.49	8.0 ± 2	1.7×10^3	-
Fresh water invertebrates	81-4681	total	T-2	13/7/81	4.1	5	85.3	19.0 ± 6	1.4×10^3	5.0 ± 2.7

decay. These plants are annual plants, obtaining their nutrients from the water soil and air each year. Levels of radionuclides in these plants are expected to reflect environmental levels on a yearly or seasonal basis. The decrease in Cs-137 activities in horsetail and iris from 1981 to 1982 suggests that an important supply of Cs-137 in these plants is the atmosphere, since this is the environmental reservoir in which the greatest decreases in Cs-137 occurred between 1981 and 1982. However, a non-linear relationship between Be-7, an atmospheric particle tracer, and Cs-137 measured in horsetail indicates other sources in addition to atmospheric deposition are important in radionuclide accumulation in horsetail (see section 6.2.2).

The variability in both Cs-137 activity and concentration factors that was present in aquatic plant samples for 1981 is again evident in the 1982 results. Cesium-137 activities in horsetail range from <4 to 63 Bq/kg ($\bar{x}$ = 20 Bq/kg) and for wild iris, Cs-137 ranges from 3-18 Bq/kg ($\bar{x}$ = 7.2 Bq/kg).

In general, Cs-137 activities and the C.F. are higher in aquatic plants by several orders of magnitude compared to values for marine plants. This is attributed to the lower ionic strength of aquatic systems where Cs-137 faces reduced competition for adsorption sites on particles and in biological systems.

The aquatic sampling program also included several animal species for radionuclide analysis. Several species of frogs, minnows (Cyprinidae), ninespine sticklebacks (Pungitius pungitius) and fresh water invertebrates (several species of worms) were analysed. Cs-137 activities in frogs ranged from 24 to 49 Bq/kg, while C.F. for Cs-137 were more uniform, 2.5 to 3.7×10^3 Bq/kg. Cs-137 activities in fish and invertebrates were lower

and Co-60 was detected in the invertebrates (Table 24). Radionuclide accumulation in amphibians and fish tends to proceed by ingestion pathways which may include uptake of bottom sediment and particulate material enriched in particle-reactive radionuclides. The importance of the ingestion process has been demonstrated in studies with fish populations where fish maintained in the water having the same activity levels had activity levels 100 times greater when fed with contaminated food over fish fed with uncontaminated food (Davis and Foster, 1958). Radionuclide content is also dependent on the size of fish, i.e. radioactivity tends to decrease with weight, but biological half-life tends to increase with size and these two factors may act to balance each other.

The monitoring program at Point Lepreau indicates that there is no concentrating effect for Cs-137 with transfer up the food chain. Williams and Pickering (1961) reported that Cs-137 concentration factors decrease as the trophic level increases from water to carnivorous fish, a view that is generally accepted (Davis, 1963).

6.2 Terrestrial Systems

Greater variability in radionuclide levels is generally observed in terrestrial compared to aquatic systems. The residence times of most radionuclides in air are much shorter than their residence times in water and, as a result, a steady state in radionuclide levels can be established throughout the different components of an aquatic system more rapidly and efficiently than throughout a terrestrial ecosystem. Physiography, microterrain and microclimate are important factors governing the nature of terrestrial ecosystems, the variables which control processes within the ecosystem and the properties of radionuclide transport within these

systems. Biological uptake of fallout radionuclides is generally more efficient in the terrestrial environment, compared to aquatic and marine ecosystems, because of the absence of a diluting medium such as water. This results in comparatively high levels of fallout radionuclides in land-based plants and animals as is described below for the baseline measurements conducted on samples collected at Point Lepreau.

6.2.1 Soil Phases

The soil component of a terrestrial ecosystem can adsorb and bind radionuclides thus serving as a sink or reservoir. Inputs result from atmospheric deposition (both dry fallout and rainfall) and decay of contaminated plants and animals. Transport mechanisms within soils can cause radionuclide migration both horizontally and vertically. The extent of lateral movement depends on the intensity and distribution of rainfall, topography and soil type. Radionuclides are removed from soils by leaching (both ground water and rainfall), adsorption by plants and animals, and by physical removal of particles (erosion). Soils are composed of a parent material, the underlying geologic or mineral substrate, and an organic component, in which organisms and their products are intermingled with the finely divided particles of the weathered parent material. The texture, porosity and its chemical composition determine the availability of soil-bound nutrients to plants and soil animals. The upper layers of soils are efficient long-term collectors of many radionuclides originating from atmospheric precipitation (Hardy et al., 1973; Nozaki et al., 1978; Dobbs and Matthews, 1976). However, physical (erosion, groundwater percolation) and biological (cultivation, animals habits, plant growth) perturbations combine to make soils a less effective recorder of radionuclide inputs than

aquatic or marine sediments.

The soil sampling program carried out in 1981 consisted of soil profiles at two sites near Point Lepreau, one at Retreat Lake and one at the very tip of Point Lepreau and a detailed discussion of results can be found in Smith et al. (1982).

The Cs-137 inventories at both locations were less than expected from the deposition from weapons fallout. The low Cs-137 inventories were probably the result of leaching and erosion processes on the soil column. The results indicate that some significant fraction of fallout radionuclides are retained by the soils in the Point Lepreau area and soil measurements can serve as useful indicators for reactor-released radionuclides.

6.2.2 Terrestrial Plants

Radionuclides are accumulated in plants by uptake from soils, direct deposition on plant surfaces or by sorption from water. Radionuclides in soluble form pass through root membranes and are subsequently incorporated into plant tissue. Radionuclides adsorbed onto plant surfaces may become incorporated into plant tissue through stomates or outer epidermal layers. In many cases, much of the radioactivity associated with plants is simply adsorbed onto surface layers and can be removed by washing or leaching (Takashima et al., 1978).

The selected species for monitoring at Point Lepreau for 1982 include lichen, moss, blueberry plants and alders. Samples were collected at sites shown in Fig. 33 and radionuclide results are tabulated in Tables 25 to 28.

Lichen

Lichen communities have long been recognized as a basic reservoir of natural and artificial radionuclides. The accumulation of radionuclides in lichen is enhanced by several factors; the persistance of aerial parts (plants with persistent aerial parts contain 2-10 times greater amounts of fallout materials than seasonal plants, (Hanson, 1966)), slow growth rate, long life span, high ratio of surface to mass and foliar absorption of nutrients. Work by Hanson et al. (1966) has shown that lichen retain approximately 10% of the fallout deposition. These morphological and physiological characteristics of lichen, which have evolved to enable these organisms to exist in environmental extremes, serve to make them an excellent indicator of contaminant fallout. Their value as fallout indicators around industrial installations is well documented and it is widely recognized that there is a qualitative and quantitative improvement in lichen flora and a decrease in pollutant content of thalli as distance increases from the source of pollution (Goyal and Seaward, 1981).

Lichen play a particularly prominent role in the arctic, where the lichen-reindeer/caribou-man food chain has been the subject of many radiological investigations. High levels of Cs-137 were detected in Alaska lichens in 1959, which was followed by the detection of high levels in caribou in 1960 (Hanson, 1966). By 1962, Hanson had measured high levels of Cs-137 in Alaskan Eskimos which were comparable to those detected in reindeer-herding Lapps from Northern Scandinavia. The reduced potassium intake that occurs when organisms utilize lichen (which has only 10 to 50 percent of the potassium concentration of higher plants) as their primary food source leads to a higher retention of Cs-137. As an example of this, caribou and reindeer which feed mainly on lichen contain up to a ten fold

higher concentration of Sr-90 and Cs-137 compared to other large herbivores such as moose, which occupy the same regions, but which prefer different foods (Holleman et al., 1971).

The radionuclide concentrations found in lichen samples for 1980, 1981 and 1982 are given in Table 25. The much higher levels of fission products in 1981 samples compared to those from 1980 and 1982 indicate that the source of these fission products is the October 1980 Chinese nuclear weapons test at Lop Nor. In July 1980, before the test, levels of short-lived radionuclides such as Ce-141, Ru-103, Ru-106, Zr-95, Nb-95 and Mn-54 were all below our detection limits. During the summer of 1981 activity levels of the fission products in lichen samples were easily detected. By the summer of 1982, the short-lived radionuclides had decayed to levels below our detection limits. The majority of lichen samples collected in 1982 contained detectable levels of only Cs-137, Ce-144 and Be-7. In a small number of samples Ru-106 ($t_{1/2}$ = 368 days) and Mn-54 ($t_{1/2}$ = 312 days) were detected. All of the short lived fission products Zr-95, Nb-95, Ce-141 and Ru-103 were below the detection limit.

The actual amount of radioactivity absorbed by a lichen community depends on several factors. The most important controlling factors include the age and size of the lichen, its efficiency at accumulating fallout radionuclides and the climatic conditions prevalent in the area. To a lesser extent, factors such as microclimate, drainage patterns and topography all play a role in absorption processes in lichen. The end result is a wide range of radionuclide concentrations for different sampling sites as was observed for lichens in the Point Lepreau area (Fig. 34). For instance, Cs-137 levels for 1982 for the ten sampling sites had an average value of 128 Bq/kg with a standard deviation of 51 bq/kg (40%). It is

Table 25. Radionuclide results for lichen samples collected near Point Lepreau

Species	Sample No.	Collection Site	Collection Date	Sample Weight	Geometry	%H2O	Cs-137 (Bq/kg)	Be-7 (Bq/kg)	Ce-144 (Bq/kg)	Ce-141 (Bq/kg)	Ru-103 (Bq/kg)	Ru-106 (Bq/kg)	Zr-95 (Bq/kg)	Nb-95 (Bq/kg)	Mn-54 (Bq/kg)
Lichen (C. rangiferina	80-280	T-1	15/7/80	113.2	1	6.2	297 ± 7	-	43 ± 13	-	-	<43	<60	-	<3.5
	80-284	T-2	15/7/80	110.5	1	6.0	222 ± 5	-	45 ± 13	-	-	<48	<73	-	<4.0
	80-290	T-3	15/7/82	119.5	1	4.6	95 ± 3	-	<33	-	-	<40	<50	-	<4.3
	80-296	T-4	7/7/80	100.3	1	4.8	185 ± 5	-	28 ± 12	-	-	<38	<80	-	<4.8
	80-303	T-6	7/7/80	103.4	1	4.6	363 ± 7	-	60 ± 20	-	-	<50	<88	-	<3.5
	81-4650	T-4	14/7/81	81.6	1	9.3	335 ± 8	583 ± 67	410 ± 33	32 ± 10	13 ± 8	<53	117 ± 8	225 ± 15	8 ± 2
	81-4651	T-3	14/7/81	95.2	1	8.8	328 ± 8	500 ± 80	322 ± 35	23 ± 12	20 ± 8	<40	93 ± 8	178 ± 19	6 ± 2
	81-4652	T-2	14/7/81	90.2	1	8.2	112 ± 5	167 ± 33	142 ± 20	9 ± 4	11 ± 5	32 ± 10	52 ± 7	88 ± 14	5 ± 2
	81-4653	T-1	14/7/81	90.4	1	8.5	156 ± 5	333 ± 50	208 ± 22	<20	15 ± 6	58 ± 20	62 ± 7	133 ± 13	4 ± 2
	81-4654	T-6	14/7/81	91.2	1	8.5	235 ± 7	550 ± 50	485 ± 32	28 ± 8	20 ± 7	117 ± 28	113 ± 8	256 ± 14	8 ± 2
	81-4655	T-7	14/7/81	77.4	1	9.0	285 ± 7	500 ± 67	333 ± 33	32 ± 10	17 ± 7	78 ± 28	97 ± 8	195 ± 18	8 ± 2
	81-4656	T-8	14/7/81	81.5	1	7.9	228 ± 7	417 ± 50	257 ± 27	14 ± 4	13 ± 6	53 ± 30	72 ± 8	157 ± 14	6 ± 2
	81-4657	T-9	14/7/81	100.1	1	9.5	167 ± 5	383 ± 33	252 ± 22	18 ± 7	13 ± 5	70 ± 22	73 ± 5	141 ± 9	6 ± 1
	81-4658	T-10	14/7/81	100.5	1	8.3	145 ± 5	283 ± 50	218 ± 20	13 ± 4	11 ± 6	<25	70 ± 7	131 ± 14	4 ± 2
	81-4659	T-11	14/7/81	73.4	1	8.5	120 ± 5	300 ± 33	200 ± 23	13 ± 7	<30	<30	50 ± 7	98 ± 10	<7
(Cladina arbuscula)	81-4665	T-7	14/7/81	45.4	1	11.7	265 ± 10	417 ± 50	393 ± 55	42 ± 23	<20	98 ± 48	105 ± 17	212 ± 42	2 ± 1
Cladonia rangiferina	82-13302	T-4	7/7/82	30.89	3	11.3	152 ± 5	149 ± 28	62 ± 18	-	-	<22	<4	-	-
	82-13303	$T-3_1$	7/7/82	82.21	2	9.14	168 ± 4	151 ± 22	64 ± 15	-	-	17 ± 13	<3	3.1 ± 1.8*	<2
	82-13304	$T-3_2$	7/7/82	74.91	2	9.40	244 ± 4	172 ± 24	78 ± 15	-	-	17 ± 14	<3	3.8 ± 1.9*	2 ± 2
	82-13305	$T-3_3$	7/7/82	118.4	1	8.89	251 ± 3	270 ± 21	111 ± 11	-	-	17 ± 11	<2	-	2 ± 1
	82-13306	$T-2_1$	7/7/82	24.75	3	9.81	76 ± 4	148 ± 36	39 ± 21	-	-	21 ± 24	<6	-	<3
	82-13307	$T-2_2$	7/7/82	23.00	3	8.96	79 ± 4	83 ± 34	29 ± 21	-	-	<25	<6	-	<3
	82-13308	$T-2_3$	7/7/82	22.05	3	8.72	77 ± 5	142 ± 43	44 ± 27	-	-	<32	<8	-	<4
	82-13309	T-1	7/7/82	78.66	2	10.20	120 ± 3	216 ± 25	53 ± 15	-	-	21 ± 13	<3	3.3 ± 2.6*	<2
	82-13310	T-6	7/7/82	26.04	3	7.35	95 ± 6	176 ± 48	52 ± 22	-	-	<27	<8	-	<3
	82-13311	T-7	7/7/82	27.76	3	17.24	149 ± 5	209 ± 43	52 ± 22	-	-	<26	<7	8.8 ± 5.2*	<3
	82-13312	T-8	7/7/82	28.11	3	15.25	101 ± 4	199 ± 42	66 ± 21	-	-	<24	<7	9.9 ± 5.0*	4 ± 3
	82-13313	T-9	8/7/82	80.22	2	10.74	92 ± 3	200 ± 28	69 ± 14	-	-	<15	<4	3.4 ± 3.3*	1 ± 1
	82-13314	T-10	8/7/82	23.47	3	23.01	74 ± 4	159 ± 34	57 ± 19	-	-	<23	<6	-	<3
	82-13315	T-11	8/7/82	31.74	3	15.10	67 ± 3	135 ± 31	34 ± 18	-	-	<21	<5	-	<2
	82-09950	T-1	3/8/82	62.30	1	11.61	167 ± 3	<534	53 ± 20	-	-	<24	<42	-	<2
	82-09951	T-1	3/8/82	75.80	1	-	124 ± 3	-	72 ± 20	-	-	<22	<38	-	3 ± 2
	82-09952	T-1	1/11/82	47.49	1	-	128 ± 4	-	102 ± 31	-	-	<35	<67	-	<4

* Not corrected for Zr-95 ingrowth.

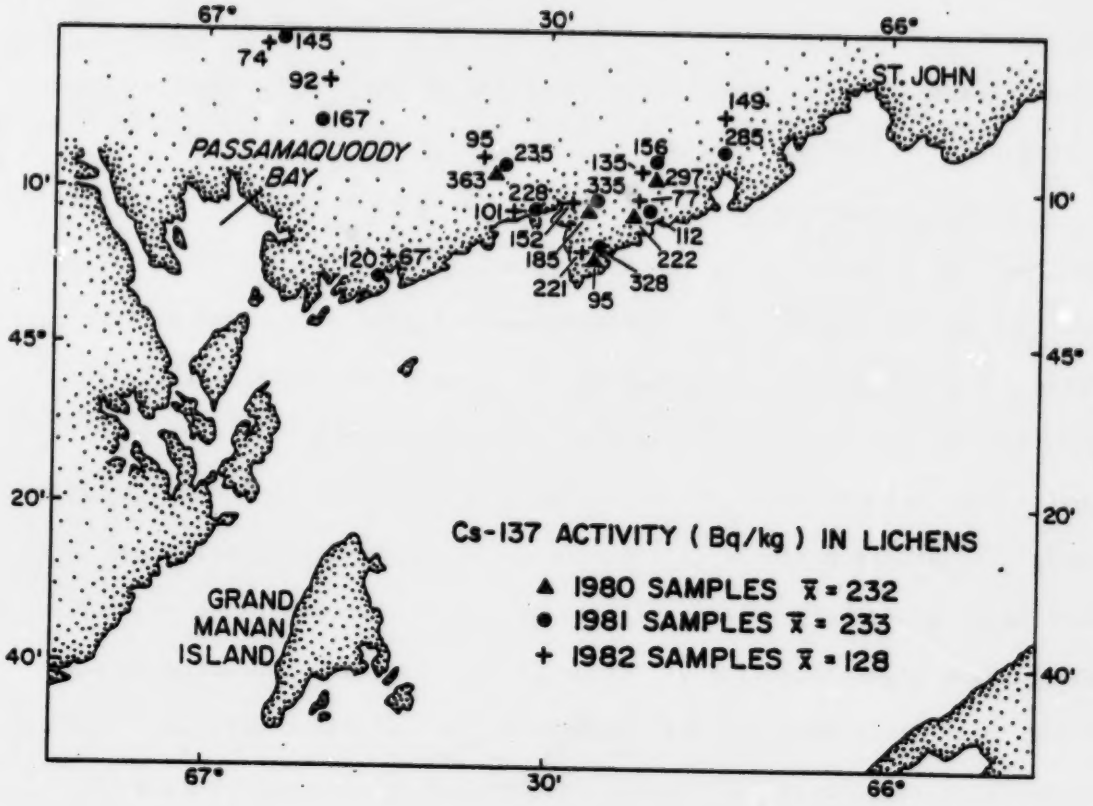


Figure 34. Distribution of Cs-137 activity levels measured in lichen for the years 1980 to 1982.

interesting to compare this variability to individual study site homogeneity, studied at site T-2 and T-3 where three replicate samples were collected. At site T-2, the average Cs-137 activity was 77 Bq/kg with a standard deviation of 1.5 Bq/kg (2.0%) indicating good site homogeneity while at site T-3, the variability was high with an average value of 221 Bq/kg and a standard deviation of 46 Bq/kg (21%). The variability in Cs-137 levels for a particular site is less than that for different sites, however variability at a particular site can be very high. Similar results were found for Be-7 and Ce-144.

This observed variability in radionuclide concentrations emphasizes the need for a normalizing factor to account for the collection efficiency (defined as the quantity of a radionuclide retained on a plant surface divided by the quantity deposited on the surface) of individual lichen plants. The short-term efficiency of each lichen sample as a sampling device for fallout radionuclides may be estimated from its concentration of Be-7, a naturally produced radionuclide. Since the atmospheric flux of Be-7 is comparatively uniform, lichen with high concentrations of Be-7 should have high collection efficiencies and thus high concentrations of other short-lived fallout radionuclides. The collection efficiency of lichen with respect to Be-7 can be estimated from its mean average air concentration of 2.4 Bq/m^3 in air monitoring samples (Table 14) collected in July 1982 and its mean concentration of 128 Bq/kg in lichen samples collected during the same period. From the ratio, Be-7 lichen activity (Bq/g)/Be-7 air concentration (Bq/m^3) = 53, it may be inferred that each gram of lichen samples approximately 53 m^3 of air during a period of several months. Therefore a 10 gram quantity of lichen samples approximately the same air volume (530 m^3) as one air monitoring station (average

vol. = 820), although the sampling time frame is not as well established for lichens.

The good correlation exhibited between Be-7 activities in lichen and those both short-lived radionuclides (Ce-144, Ru-106, etc) and long-lived radionuclides (Cs-137) can be used to gain insight into the radionuclide collection behavior of lichen. The 1981 lichen results (Smith et al., 1982) exhibited linear relationships between Be-7 and both the short-lived radionuclides, as well as Cs-137 with correlation coefficients close to 0.9. These results indicate that collection efficiencies of lichen samples remain constant for the entire life time of the radionuclides. The linear relationships observed between Cs-137, Mn-54 and Ce-144 and Be-7 in 1981 lichen samples are still observed when the 1982 samples (all decay corrected to 1981) are included (Fig. 35).

The Cs-137 content of lichen samples collected in 1982 was significantly lower than average values for 1980 and 1981. Fig. 34 illustrates the Cs-137 distribution at various sampling sites near Point Lepreau for 1981 and 1982. As illustrated, the 1982 average Cs-137 value of 128 Bq/kg is much lower than the values of 232 Bq/kg in 1980 and 233 Bq/kg in 1981. Despite the wide variability in Cs-137 values from site to site which reflects environmental heterogeneity, a linear relationship is maintained between Cs-137 and Be-7. This indicates that Be-7 accurately reflects the extent of the lichen exposure to atmospheric particulate material.

Cerium-144 ($t_{1/2}$ = 284d) and Cs-137 ($t_{1/2}$ = 30y) are two radionuclides consistently measured in lichen during the pre-operational phase of PLEMP. The measured ratios of Ce-144/Cs-137, shown in Fig. 36 for 1980, 1981 and 1982, are characterized as follows; in the summer before the October, 1980 Chinese nuclear test, Ce-144 and Cs-137 were measured at an

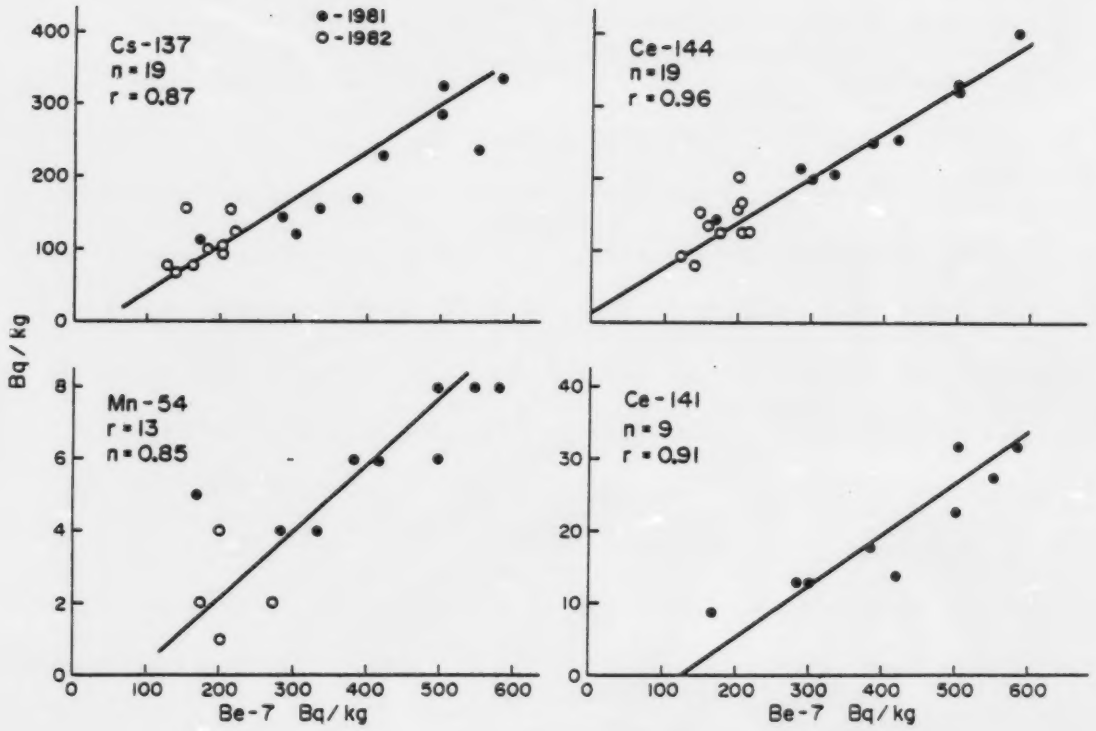


Figure 35. Relationship between fallout radionuclide levels in lichen and their Be-7 activity. Arrows indicate values below the detection limit. Least squares linear fit having high correlation coefficient (r) in most cases indicates that Be-7 is a good calibration radionuclide for estimating the collection efficiency of lichen for a broad range of radionuclides.

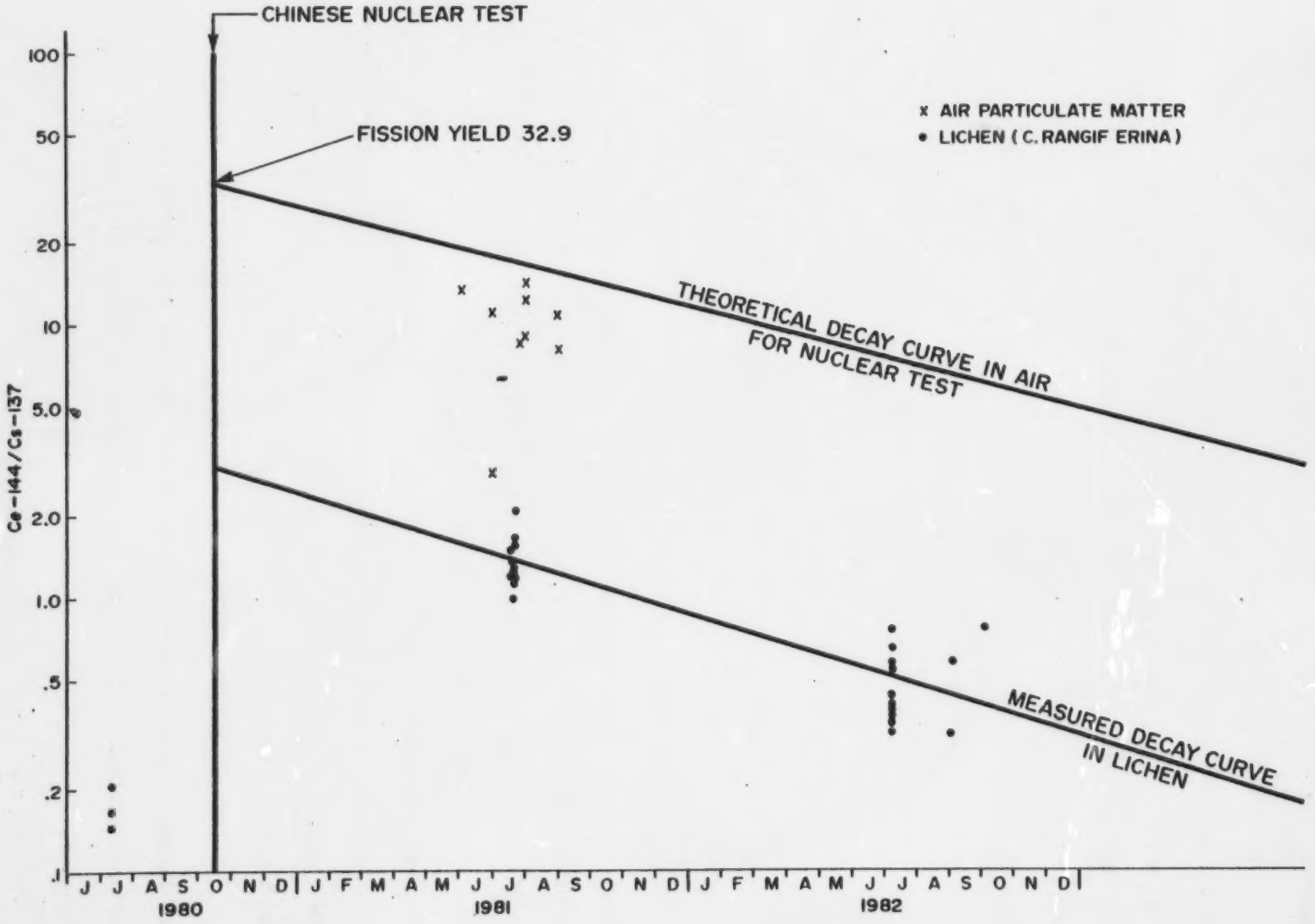


Figure 36. Measured isotopic ratio of Ce-144/Cs-137 for air particulate material and lichen 1980 to 1982. Solid line is the theoretical decay ratio for the Chinese nuclear test (Oct. 1980) assuming an initial fission yield of 8.4 for this isotopic ratio.

average ratio of 0.17 (std. dev., $\sigma = 0.03$). Samples collected in the summer of 1981 had an average ratio of 1.4 ($\sigma = 0.3$) reflecting the recent input of Ce-144 with respect to Cs-137 from the nuclear test. The ratio was smaller than that expected strictly from atmospheric deposition suggesting that a reservoir of Cs-137 is present in lichen due to the long life of the lichen (Smith et al., 1982). Ratios measured for samples collected in 1982 have lower ratios (average ratio of 0.50, $\sigma = 0.14$). The decrease in ratio between 1981 and 1982 is that predicted by the radioactive decay of Ce-144 and Cs-137 for this time period so that Ce-144 or Cs-137 are not removed or retained preferentially over the other by lichen over the time period of a year.

It has been suggested that Cs-137 uptake in plants is inversely proportional to the exchangeable potassium in the soil (Anderson et al., 1957). Fig. 37 shows Cs-137 plotted against K-40 for lichen samples collected in the Point Lepreau area for 1980-82. The Cs-137 content of lichen exhibits the predicted inverse relationship to the K-40 content ($r = 0.6$, $n = 26$). This relationship holds for samples collected both before and after the 1980 Chinese weapons test. Thus, it seems that the two controlling factors governing the distribution of Cs-137 in lichen are; (1) the retention efficiency of lichen for fallout radionuclides, as indicated by the Be-7 activity and, (2) the quantity of exchangeable potassium competing with Cs-137 for uptake.

In general, the linearity of each radionuclide relationship to Be-7 in lichen, and the sensitivity of lichen radionuclide levels to the 1980 Chinese nuclear test indicate the usefulness of lichen as a monitoring species.

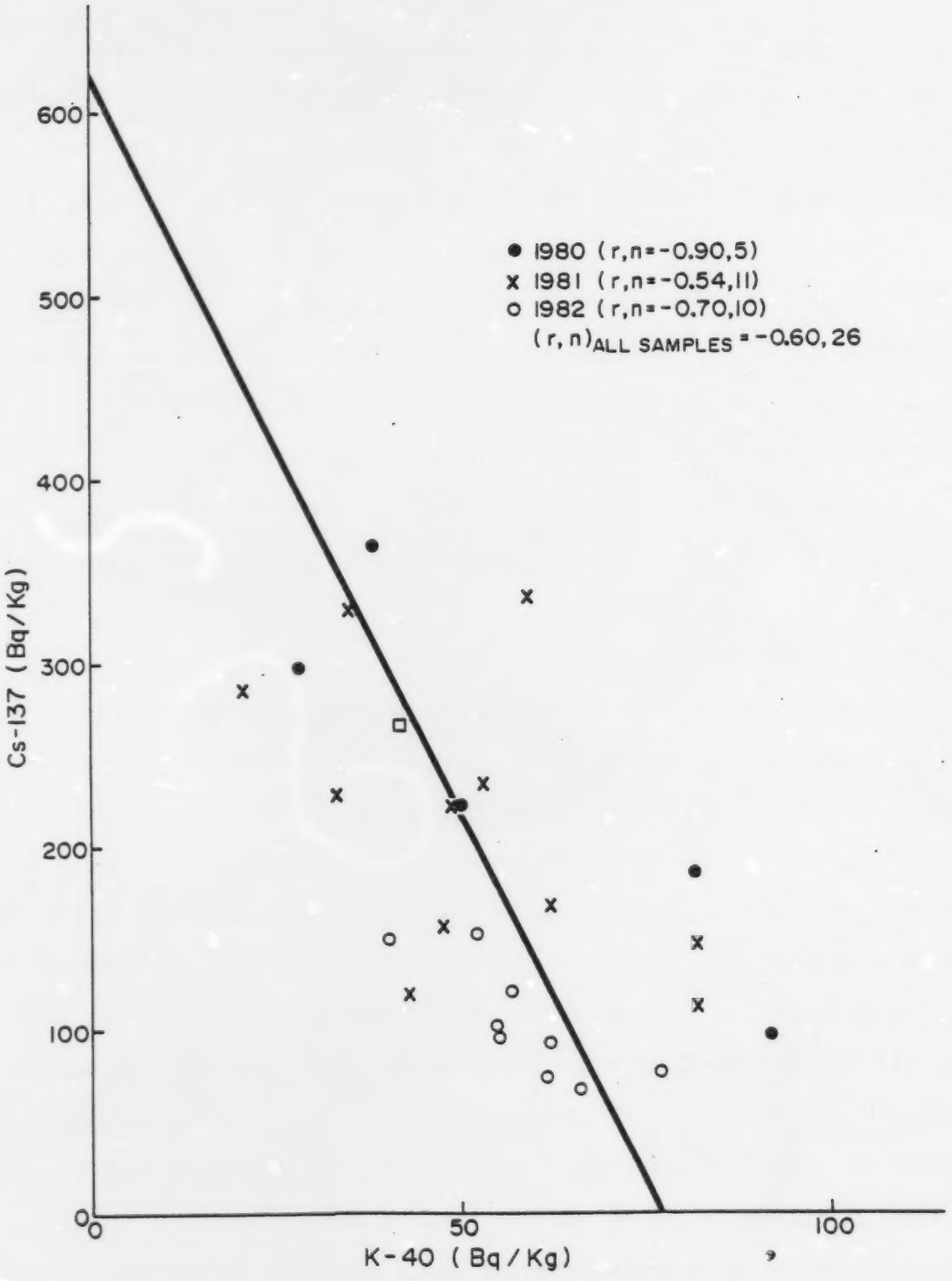


Figure 37. Cesium-137 activity levels as a function of the K.40 content for samples collected in 1980 to 1982. The inverse, linear relationship has a correlation coefficient of -0.60.

Moss, Alder and Blueberry Leaves

Sphagnum moss generally grows in a compact mat and because of its high cation exchange capacity acts as a sink for metal ions in a terrestrial ecosystem. Sphagnum bogs are efficient collectors of fallout and tend to assimilate radionuclides in the dissolved form. Cs-137 levels on the order of 200 Bq/kg have been reported (Svoboda and Taylor, 1979) in Sphagnum samples collected in Vermount, Ontario and the North West Territories in the early 1970's. Mosses are useful indicator species, providing a good measurement of radioactivity levels circulating through the terrestrial hydrological system.

Radionuclide results for moss samples collected in 1980 and 1982 are given in Table 26. Unfortunately, no samples were collected in 1981 for which levels of fission products from the October 1980 nuclear weapons test could have been determined.

A wide range of Cs-137 activities was found between different sites with activities ranging from 24 to 165 Bq/kg in 1980 and from 4 to 930 Bq/kg in 1982. This variability is expected in mosses since radionuclide uptake occurs from solution and uptake rates are strongly influenced by the rainfall and the drainage rate for soils. Samples collected in 1982 were higher in Cs-137 (an average activity of 265 Bq/kg in 1982 compared to 74 Bq/kg in 1980). This is the only plant sampled where an increase in Cs-137 activity was observed between 1980 and 1982. This increase can only be explained as a result of increases in Cs-137 activities in the atmosphere as a result of the Chinese nuclear test since activities did not increase significantly in the sediment, soil or water reservoirs. Atmospheric deposits are therefore a major source of Cs-137 for mosses. The greater effect of the Chinese nuclear test on fallout levels in moss

Table 26. Radionuclide results for moss samples collected near Point Lepreau.

Species	Sample No.	Collection Site	Collection Date	Sample Weight	Geometry	%H_2O	Cs-137 (Bq/kg)	Be-7 (Bq/kg)	Ce-144 (Bq/kg)	Ru-106 (Bq/kg)	Zr-95 (Bq/kg)	Mn-54 (Bq/kg)
Bog Moss (Sphagnum sp.)	80-275	T-1	1/12/80	38.7	1	75.0	30 ± 7	-	< 73	<70	< 55	<7
	80-282	T-2	3/12/80	39.7	1	74.5	93 ± 7	-	< 70	<62	< 50	<7
	80-285	T-3	1/12/80	32.0	1	79.7	58 ± 7	210 ± 120	< 82	<77	< 53	<5
	80-293	T-4	3/12/80	32.0	1	74.7	165 ± 5	-	30 ± 15	<50	< 98	<5
	80-299	T-5	3/12/80	19.1	1	87.2	24 ± 7	-	< 80	<55	< 70	<5
	82-13335	$T-1_1$	7/7/82	44.31	1	59.21	469 ± 8	64 ± 71	33 ± 24	<28	< 7	<3
	82-13336	$T-1_2$	7/7/82	61.74	1	46.72	930 ± 11	113 ± 98	62 ± 28	<32	< 8	<3
	82-13337	$T-1_3$	7/7/82	28.79	2	58.80	299 ± 5	99 ± 65	28 ± 24	<24	< 9	<3
	82-13338	T-2	7/7/82	29.61	1	-	157 ± 8	<3900	< 70	<82	<296	<6
	82-13339	$T-3_1$	7/7/82	23.13	2	45.94	75 ± 7	108 ± 106	< 51	<50	< 19	<6
	82-13340	$T-3_2$	7/7/82	45.82	1	40.99	121 ± 1	-	-	-	-	-
	82-13341	$T-3_3$	7/7/82	23.00	2	48.81	243 ± 15	-	<190	-	-	-
	82-13342	T-4	7/7/82	37.45	1	76.88	4 ± 4	<2400	< 42	<71	<179	<7
	82-13343	T-4	7/7/82	38.57	1	23.56	176 ± 5	<1500	92 ± 44	<49	<122	<5

compared to lichen is reflected in relatively greater 1982 to 1980 Cs-137 levels in moss. One possible explanation is that moss ages are less than those for lichen resulting in an enhanced sensitivity to recent fallout phenonema.

A poor correlation is observed between Cs-137 and Be-7 in moss ($r=-0.42$, $n=5$) which is a little surprising in view of the importance of atmospheric deposition to levels found in moss. This anomaly may be explained by considering the method of radionuclide assimilation by mosses. Radionuclides are incorporated by mosses from the dissolved phase and are therefore strongly affected by surface water concentrations. Atmospheric inputs and the removal rates of radionuclides from the moss will vary in a manner related to the amount of rainfall and subsequent processes which affect the concentration of radionuclides in surface water. Non-steady state effects associated with variable rainfall, fractionation between Be-7 and Cs-137 in surface water, ground water and the differential uptake of these radionuclides contribute to the non-linear relationship between Be-7 and Cs-137 levels in moss.

Ce-144 was detected in most of the 1982 moss samples. A comparison with activities found in 1980 is not possible since all activities except one in 1980 were below the relatively high detection limit. The one sample which had a measurable activity (30 Bq/kg) lies in the range of samples measured in 1982 (28-92 Bq/kg).

Alnus rugosa (speckled alder) is found predominantly in drainage areas and consequently it is a good indicator for water-borne radionuclides. Alder is not a primary browse species, but is typical for plants at the base of the food web and is utilized by insects which are themselves consumed by amphibians, birds, mammals and reptiles. Table 27 shows the

Table 27. Radionuclide results for alder collected near Point Lepreau.

Sample	Sample No.	Type	Collection Site	Collection Date	Sample Wt (g)	Sample Geometry	%H_2O	% Ash	Cs-137 (Bq/kg)	Other Radionuclides (Bq/kg)
Alder (Alnus rugosa)	80-279a	branch (ash)	T-1	1/12/80	9.1	4	-	0.98	3.18 ± 0.07	
	80-279b	buds	T-1	1/13/80	13.4	4	-	-	26.7 ± 3.3	
	80-283a	branch	T-2	3/12/80	66.6	2	-	-	5.5 ± 1.7	Ce-144 (4.6 ± 1.0)
	80-283a	branch (ash)	T-2	3/12/80						
	80-289a	branch (ash)	T-3	1/12/80	10.7	4	-	1.0	4.4 ± 0.2	
	80-302a	branch (ash)	T-5	3/12/80						
	80-283b	buds	T-2	3/12/80						
	80-289b	buds	T-3	1/12/80	34.9	3	-	-	15.7 ± 1.8	
	80-295b	buds	T-4	3/12/80						
	80-302b	buds	T-5	3/12/80						
	80-295a	branch	T-4	3/12/80	20.9	3	-	-	3.2 ± 1.7	
	80-302a	branch	T-5	3/12/80	16.3	3	-	-	4.5 ± 2.5	
	82-13344	leaves	T-1	7/7/82	47.72	1	63.64	-	21 ± 2	Be-7 (78 ± 26)
	82-13345	leaves	T-2	7/7/82	56.75	1	56.28	-	27 ± 2	Be-7 (52 ± 35)
	82-13346	leaves	T-3	7/7/82	61.90	1	45.69	-	<2	Be-7 (73 ± 46)
	82-13347	leaves	T-4	7/7/82	47.48	1	62.80	-	35 ± 2	Be-7 (70 ± 32)
	82-13348	leaves	T-5	8/7/82	57.55	1	67.15	-	3.6 ± 0.6	

radionuclide results for alder samples collected near Point Lepreau for 1980 and 1982. The results show that Cs-137 levels in leaves and buds are similar and approximately an order of magnitude higher than Cs-137 levels found in the branches which agrees with work by Olson and Cate (1960) who, using decidious trees, have shown that Cs-134 is concentrated in buds and leaves after injection into tree bore holes.

Results for blueberry plants at Point Lepreau are given in Table 28. The leaves show an affinity for Cs-137 and generally contain much higher Cs-137 levels than alder samples collected from the same location. For example, levels of Cs-137 in 1982 at site T-1 are 51 Bq/kg and 21 Bq/kg and at site T-2 are 109 Bq/kg and 27 Bq/kg for blueberry leaves and alder leaves respectively.

Cs-137 levels are quite variable and range from 15 Bq/kg to 197 Bq/kg for 1982. Similar observations were made by Remmenga and Whicker (1967) for a single drainage system in which Cs-137 activities in blueberry plants varied from 30 to 300 Bq/kg. They also observed, as we did between alder and blueberry leaves, considerable variation between species growing in the same sampling sites, eg. blueberry contained four to seven times the Cs-137 activity of aspen from sampling sites 10 m apart. The uptake of radionuclides depends on several factors including degree of fallout entrapment, plant surface characteristics (smooth vs rough), age of plant, environment, inventory and availability of individual radionuclides to the root system and the physiological demand for the radionuclide or stable analogue. These factors lead to the wide inter- and intra-species variation in the uptake of radionuclides in most aquatic and terrestrial plants.

The uptake of radionuclides by blueberry and alder leaves, like the horsetail follows a more complex physio-chemical process than uptake by

Table 28. Radionuclide results for blueberry leaves collected near Point Lepreau.

Sample No.	Collection Site	Collection Date	Sample Weight	Geometry	%H_2O	Cs-137 (Bq/kg)	Be-7 (Bq/kg)	Ce-144 (Bq/kg)	Ce-141 (Bq/kg)	Ru-103 (Bq/kg)	Ru-106 (Bq/kg)	Zr-95 (Bq/kg)	Nb-95 (Bq/kg)	Mn-54 (Bq/kg)
81-4660	T-4	14/7/81	71.9	1	8.2*	253 ± 7	80 ± 23	130 ± 23	7 ± 3	<8	<30	13 ± 6	41 ± 13	<3
81-4661	T-3	14/7/81	46.2	1	7.6*	103 ± 7	97 ± 40	88 ± 37	<28	<15	<48	10 ± 6	38 ± 24	<5
81-4662	T-2	14/7/81	50.2	1	5.7*	56 ± 5	72 ± 28	89 ± 28	<17	15 ± 6	<38	23 ± 6	56 ± 20	4 ± 2
81-4663	T-1	14/7/81	35.9	1	5.1*	58 ± 7	102 ± 47	205 ± 50	<35	<19	<67	42 ± 12	70 ± 48	5 ± 3
81-4664	T-7	14/7/81	43.8	1	5.4*	15 ± 3	97 ± 45	77 ± 22	<16	<9	<20	35 ± 7	46 ± 19	5 ± 2
81-4666	T-10	14/7/81	26.0	1	4.8*	83 ± 8	145 ± 47	90 ± 30	<50	<23	<80	83 ± 33	-	5 ± 3
81-4667	T-12	14/7/81	30.0	1	5.3*	102 ± 5	160 ± 80	110 ± 35	<28	<14	<45	50 ± 12	38 ± 36	5 ± 2
82-13321	T-4	7/7/82	32.14	1	63.90	51 ± 4	46 ± 44	<23	-	-	<25	< 8	-	<3
82-13322	T-3	7/7/82	36.67	1	61.07	109 ± 5	< 52	<24	-	-	<28	< 8	-	<3
82-13323	T-3	7/7/82	30.56	1	63.30	74 ± 5	< 61	<28	-	-	<32	< 10	-	<4
82-13324	T-3	7/7/82	40.01	1	62.32	106 ± 4	< 50	<21	-	-	<24	< 7	-	<3
82-13325	T-2	7/7/82	31.11	1	60.17	37 ± 2	40 ± 29	<14	-	-	<15	< 5	-	<2
82-13326	T-2	7/7/82	31.61	1	63.28	51 ± 4	73 ± 50	<25	-	-	<29	< 9	-	<3
82-13327	T-2	7/7/82	42.75	1	64.34	84 ± 4	< 49	<20	-	-	<23	< 8	-	3 ± 3
82-13328	T-1	7/7/82	35.10	1	60.77	28 ± 3	< 53	<23	-	-	<26	< 9	-	<3
82-13329	T-5	7/7/82	47.60	1	63.37	16 ± 3	<1500	<33	-	-	<38	<146	-	3 ± 3
82-13330	T-7	7/7/82	43.74	1	61.61	184 ± 4	<1700	<30	-	-	<36	<113	-	3 ± 2
82-13331	T-8	7/7/82	41.23	1	60.50	197 ± 5	<1400	<30	-	-	<37	<106	-	3 ± 2
82-13332	T-9	8/7/82	37.45	1	64.32	15 ± 3	-	<40	-	-	<45	-	-	<5
82-13333	T-10	8/7/82	49.00	1	58.37	36 ± 3	<1000	<28	-	-	<34	<109	-	<4
82-13334	T-12	8/7/82	40.90	1	64.20	29 ± 1	-	-	-	-	-	-	-	-
82-09953	T-10	31/8/82	36.04	1	61.55	76 ± 4	< 900	<34	-	-	-	< 82	-	<7
82-09954A	T-12	31/8/82	41.08	1	63.27	14 ± 2	-	-	-	-	-	-	-	-
82-09954B	T-12	31/8/82	42.21	1	63.27	15 ± 2	-	-	-	-	-	-	-	-
82-09954C	T-12	31/8/82	31.75	1	63.27	15 ± 1	-	-	-	-	-	-	-	-

* 1981 blueberry leaves received in semi-dry form

lichen. The relationship between Cs-137 and Be-7 (Fig. 38), is non-linear and is not a simple function of the quantity of atmospheric particles deposited. In general, blueberry plants are effective in accumulating fallout radionuclides (short-lived radionuclides such as Ce-141, Ce-144, Zr-95 etc were detected in 1981) but the data is less reliable, shows more variance and is harder to interpret than those for the lichen.

6.2.3 Aquatic and Terrestrial Mammals

The large variations in radionuclide concentrations measured among different species of vegetation points to the importance of food habits in the accumulation of radionuclides by herbivores. Many of the studies concerning radioactivity uptake by herbivores have focused on deer as these are important agents for radionuclide transport to carnivores and a potential pathway to humans (Whicker et al., 1965; Hanson et al., 1963; Longhurst et al., 1967). Measurements of Cs-137 in mule deer collected in Colorado between 1962 and 1965 gave values ranging from 7.4 to 150 Bq/kg (Whicker et al., 1965) which followed a distinct seasonal pattern. Deer flesh in Colorado in the mid-1960's contained five to ten times the concentrations of Cs-137 measured in beef or pork. It was also noted that the consumption of deer liver could lead to appreciable uptake of Ce-144, Cs-137, Mn-54, and Ru-106, all of which are concentrated in this organ. The transfer of Cs-137 from mule deer to a principal predator, the mountain lion (Felis concolor) has received some study and it has been noted that a three fold increase in the Cs-137 concentration accompanies this transfer (Pendleton et al., 1964).

The three principal herbivores monitored for radioactivity at Point Lepreau are the muskrat (Ondatra zibethicus), varying hare (Lepus

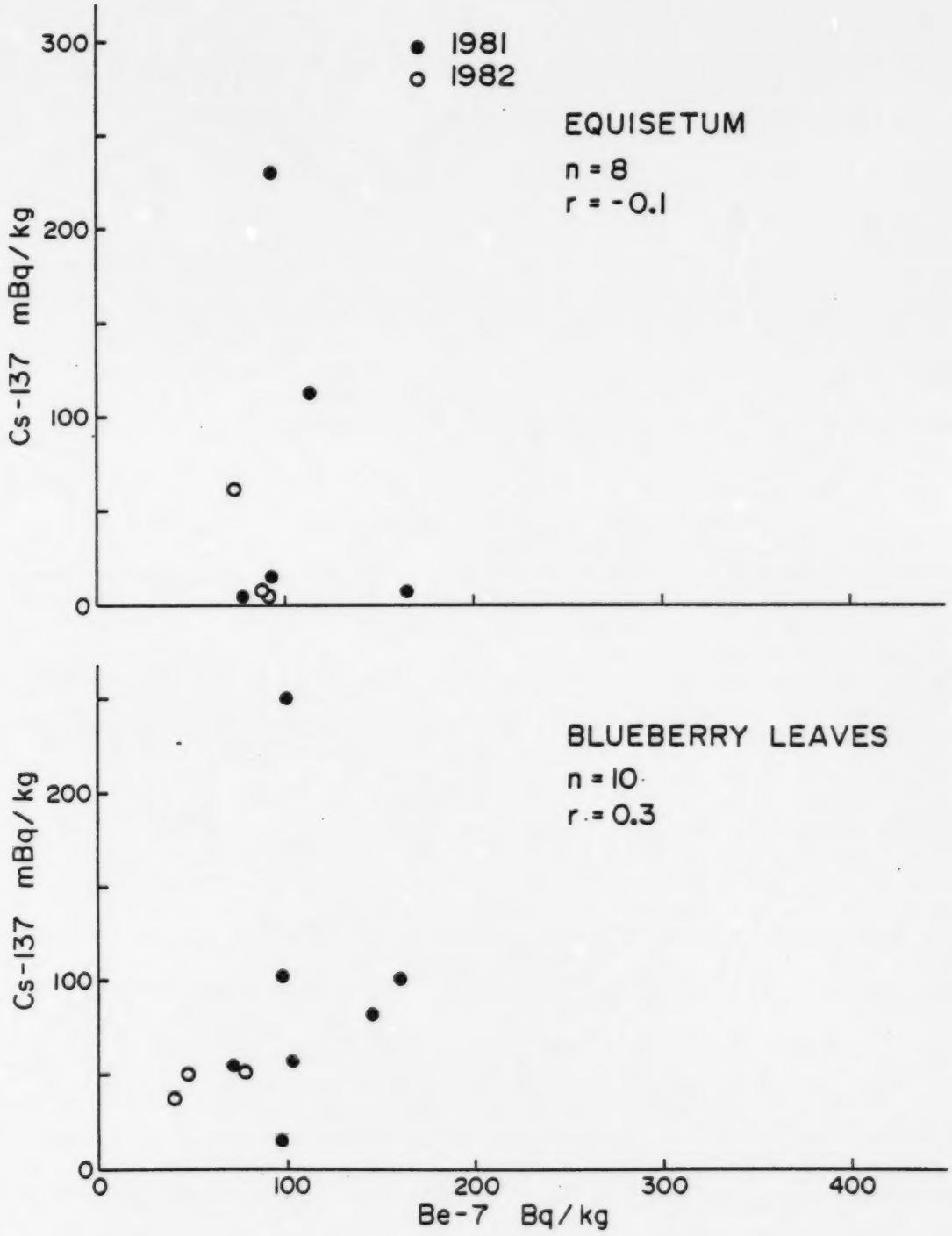


Figure 38. Relationship between Cs-137 activity and Be-7 activity in equisetum indicates that processes other than simple physical sorption may be significant in radionuclide uptake.

americanus) and meadow vole (Microtus pennsylvanicus). One sample of liver of the white-tailed deer (Odocoileus virginianus) was also examined in 1982. Radionuclide results for aquatic and terrestrial mammals for 1980 and 1982 are given in Table 29 (site locations are shown in Fig. 33). Cs-137 was measured at a level of 49 Bq/kg in deer liver and 132 Bq/kg in kidney. This value is similar to levels found in hare and muskrat liver and in meadow vole. The level of Cs-137 in the deer liver and kidney is within the range found in blueberry leaves in 1981 (Table 29), which are typical food for deer. There appears to be no enrichment of Cs-137 in this transfer from plants to deer.

Cesium-137 activities measured in muskrat and hare samples collected in 1982 are much lower than for those collected in 1980 (Table 29). A similar decrease in Cs-137 levels was observed for aquatic and terrestrial plants, known to the food of these animals, from 1981 to 1982 and probably accounts for the decrease in Cs-137 activity in the animals. The increase in variability, observed particularly for muskrat, is a function of increased number of samples.

The highest Cs-137 levels in muskrats collected at Point Lepreau were measured in the muscle tissue (Table 29). This result is in agreement with previous work (Ballou and Thompson, 1958) which has indicated that the voluntary muscle of terrestrial mammals is generally the dominant accumulator of cesium, both on total body and per gram tissue basis. Nelson et al. (1961) conducted a study in which Cs-137 uptake was traced throughout the tissues of mice. The measured distributions 36 days after the administration of the tracer was approximately consistent with the results for muskrat given in Table 29 with the highest levels measured in the muscle and the lowest measured in bone. It is interesting to note that the Cs-137

Table 29. Radionuclide results for vertebrate animals collected near Point Lepreau.

Organism	Sample No.	Organ	Collection Site	Collection Date	Sample Weight (g)	Sample Geometry	%H_2O	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Co-60 (Bq/kg)
Muskrat (Ondatra zibethicus)	80-297a	liver	T-4	31/10/80	26.53	3	72.4	108 ± 5	1.5×10^4	<4.3
	80-304a	liver	T-5	07/11/80	8.64	5	72.7	53 ± 8	7.2×10^3	<3.3
	82-13361	liver	T-1	15/03/82	9.51	4	61.6	<3	<580	<3.0
	82-9940	liver	T-2	15/03/82	4.59	5	73.1	<4.9	<660	<6.1
	82-9960	liver*	T-2,4,5	15/11/82	19.93	4	74.1	19 ± 4	2.5×10^3	<3.2
	80-297b	muscle	T-4	31/10/80	6.84	4	74.0	173 ± 15	2.3×10^4	<6.7
	80-304b	muscle	T-5	07/11/80						
	82-13362	muscle	T-1	15/03/82	17.0	3	69.8	14 ± 7	2.1×10^3	<4.3
	82-9943	muscle	T-2	15/03/82	24.69	3	73.0	3.3 ± 1.6	445	<2.2
	82-9955	muscle	T-1	15/11/82	26.88	3	75.9	1.9 ± 0.7	229	<1.1
	82-9959	muscle	T-2	15/11/82	11.05	4	72.9	105 ± 9	1.4×10^4	
	82-9967	muscle	T-4	15/11/82	27.55	3	73.8	2.5 ± 1.2	328	<1.7
	82-9971	muscle	T-5	15/11/82	21.50	3	76.5	8.6 ± 3.0	1.0×10^3	<2.1
	80-297c	bone	T-4	31/10/80						
	80-304c	bone	T-5	07/11/80	10.04	5	37.0	5 ± 1.5	1.7×10^3	2.2 ± 1.5
	82-9952	bone*	T-1,2,4,5	15/11/82	16.87	4	28.6	<3	$<1.0 \times 10^3$	<3.5
	80-297d	brain	T-4	31/10/80				77 ± 25	1.1×10^4	23 ± 12
	80-304d	brain	T-5	07/11/80	1.33	5	72.0			
	82-9939	brain	T-1,2,4,5	15/11/82	4.29	5	76.8	<4.8	<5.60	<6.1
	82-9937	stomach	T-4	15/11/82	12.72	4	76.6	<1.5	<180	<2.2
	82-9938	stomach*	T-1,2,5	15/11/82	18.25	3	81.2	3.8 ± 1.7	360	<2.8
Deer	81-13359a	liver	T-5	15/08/81	185.0	1	64.9	48.8 ± 0.8	8.5×10^3	-
	81-13359b	kidney	T-5	15/08/81	80.05	2	64.9	132 ± 10	2.3×10^4	<2.9
Varying Hare (Lepus americanus)	80-305a	liver	T-5	07/11/80	20.42	3	76.3	283 ± 7	3.4×10^4	2.5 ± 2.0
	80-305b	muscle	T-5	07/11/80	5.45	4	79.4	410 ± 17	4.2×10^4	6.7 ± 5.2
	80-305c	brain, lung, kidney	T-5	07/11/80	5.89	4	80.5	80 ± 3	7.8×10^3	<3.3
	82-9946*	muscle	T-1	15/11/82	24.76	3	75.4	50 ± 2	6.1×10^3	<1.2
	82-9948*	brain	T-1	15/11/82	1.52	5	79.0	<19	$<2.0 \times 10^3$	<24
Meadow Vole (Microtus pennsylvanicus)	80-306	muscle + liver + intestine	T-5	07/11/80	2.66	5	84.2	37 ± 10	2.9×10^3	7.7 ± 4.3

* Composite bone sample.

concentration factors for muskrat tissues are approximately the same order of magnitude as those for the aquatic plants which they are known to consume (Table 23). Hence, there is no obvious concentration or discrimination effect in Cs-137 transport between plants and muskrats. However, the Co-60 concentration in brain tissue (23 Bq/kg) is much higher than levels measured either in vegetation or other tissues of the muskrat. This organ exhibits a marked concentrating effect for cobalt.

The Cs-137 levels in the varying hare were greatest in the muscle tissue, as well, approximately twice as great as levels measured in muskrats muscle collected during the same time period. The slightly higher levels of Cs-137 in hares compared to muskrats may reflect the slightly greater concentrations of this radionuclide in terrestrial plants compared to aquatic plants. Both Cs-137 and Co-60 were easily detected in the meadow vole ('field mouse') at levels similar to those measured in both aquatic and terrestrial vegetation.

Cesium-137 concentrations in aquatic and terrestrial plants and in the muscles of herbivores, which tend to feed on these plants, are of the same order of magnitude (Table 30). Hence, there is very little change in Cs-137 activities during transport through this link in the food chain. However, there is a great difference in Cs-137 levels between the different tissues of terrestrial mammals and the same tissues in a marine mammal (harbour seal; Table 31). Cesium-137 levels in terrestrial mammals are approximately two orders of magnitude greater than the levels in harbour seals. This is consistent with the previous observation that Cs-137 levels in aquatic plants and fish are also two orders of magnitude greater than levels in marine plants and fish.

Table 30. Cs-137 in terrestrial samples for 1980-1982.

Sample	Activity Cs-137 (Bq/Kg) 1980-81 Range	1980-81 Average	1982 Range	1982 Average
Horse tail	5-233	62	<2 - 63	16
Lichen	95-363	233	67 - 251	180
Sphagnum	58-165	99	4 - 930	320
Blueberry Leaves	15-253	96	14 - 197	67
Deer Liver	-	50	-	-
Deer Kidney	-	132	-	-
Muskrat Muscle	-	183	-	14
Hare Muscle	-	410	-	50
Meadow Vole	-	37	-	-
Soil (A Horizon)	10-80	-	-	-

Table 31. Comparison of Cs-137 activities in terrestrial and marine mammals.

	Terrestrial						Marine
Organ	Muskrat		Hare		Meadow vole(1)	deer(1)	Harbour Seal(3)
	1980(3)	1982(6)	1980(1)	1982(1)	1980	1981	1980
Muscle	173	14	410	50	-	-	1.5
Liver	82	12	283	-	-	49	1.6
Kidney	-		-		-	132	-
Brain	77	<4.8	80*	<19		-	
Total	-	-	-		37	-	1.5
% Moisture	71	75	79	77	84	65	70

* Includes kidney

6.3 Tritium in Aquatic and Terrestrial Systems

The results for tritium in aquatic and terrestrial systems are presented in Table 32 and levels found were within the range of tritium activity found for freshwater samples in the region (4.2 Bq/l to 10.6 Bq/l). Aquatic mammals (i.e. Ondrata zibethicus, muskrat) had the highest levels of tritium, while terrestrial plants (i.e. Alnus rugosa, alder, and C. Rangiferna, lichen) exhibited the lowest levels of tritium observed. Concentration factors ranged from 0.3 to 1.1 and followed trends observed previously in the literature. Harrison et al. (1973) found that both cattails Typha angustifolia and filamentous algae (Pithophora sp.) raised in a contaminated pool reached tritium levels from 70 to 100% of the original pool water but no concentration effects were observed. Adams et al. (1979) worked with an experimentally contaminated freshwater marsh ecosystem and found that tritium levels in submergent aquatic plants were higher than for emergent plants but in both cases concentration factors were approximately one or less. Adams found that for both plants and animals tritium levels generally followed marsh water tritium levels. Our results are in excellent agreement and tritium levels measured for all plant and animal species fall within the range of tritium levels measured for various freshwater systems in the Point Lepreau region.

6.4 Birds

In general, birds are not particularly suited for use as indicator species for radionuclide uptake because of their mobility over a comparatively wide range. On the other hand, they do offer several advantages as environmental monitors; resident species remain in a fixed area of known extent, predictable influxes of migrant populations provide a new sample

Table 32. Tritium levels found in aquatic and terrestrial systems.

Species		Sample No.	Site No.	H-3 (Bq/l)	Concentration Factor
Water		avg. 7.54 Bq/l			
Plants					
Equisetum sp.		82-13357	T-4	4.5 ± 1.6	0.6
Iris versicolor		82-13353	T-5	5.4 ± 1.6	0.7
C. Rangiferna		82-9951	T-1	3.6 ± 2.2	0.5
		82-9952	T-1	<1.8	-
Vaccinium Sp. (Leaves)		82-9953	T-10	4.3 ± 2.2	0.6
(Blueberry)		82-9954	T-12	2.8 ± 2.2	0.4
		82-13333	T-10	5.4 ± 1.6	0.7
Sphagnum Sp.		82-13338	T-2	6.0 ± 1.6	0.8
Alnus rugosa		82-13348	T-5	3.7 ± 1.6	0.5
Animals					
Ondatra zibethicus					
(muskrat)	muscle	82-13366	T-2	8.4 ± 1.8	1.1
		82-9955	T-1	5.4 ± 1.8	0.7
		82-9959	T-2	7.8 ± 1.9	1.0
		82-9967	T-4	8.1 ± 1.9	1.1
	liver	82-13365	T-2	5.1 ± 1.8	0.7
		82-9956	T-1	3.0 ± 1.8	0.4
	stomach	82-9936	T-2	2.4 ± 1.8	0.3
Odocoileus virginianus (Deer)	liver	82-13359	T-2	4.6 ± 1.6	0.6
Lepus americanus (Hare)	liver	82-9947	T-1	3.6 ± 1.8	0.5
	stomach	82-9949	T-1	4.3 ± 1.8	0.6

not in equilibrium with the study area, their high metabolism makes them good bio-accumulators, and it is possible to sample distinct trophic levels within the food web. Work by Levy et al. (1983) has shown that radionuclide levels in birds were for the most part dependent on the tropic level that each species occupied in the same local habitat. Uptake of radionuclides occurs by sorption from food or other ingested materials or by direct adsorption.

Twenty-one artificial radionuclides have been measured by Markham and Halford (1982) in mourning doves in the Idaho National Engineering Laboratory (INEL) Site in southwest Idaho. Highest activities found were for the radionuclides Cs-137, Cr-51, Co-60, I-131 and Ce-144. Activities of Cs-137 of up to 6300 Bq/kg dry weight in muscle and up to 16,000 Bq/kg in gastrointestinal tracts were measured. Background levels of Cs-137 were generally below 22 Bq/kg. Willard (1960) observed that Cs-137 levels in bird muscle increased dramatically in winter as a result of ingestion by birds of contaminated soil while foraging for food.

The birds collected for monitoring purposes at various sites near Point Lepreau (Fig. 39) are, for the marine environment: (1) Phalocrocorax carbo (great cormorant), Somateria mollissima (eider duck), Larus argentatus (herring gull) and Calidris maritima (purple sandpiper); and for the terrestrial environment; Sturnus vulgaris (starling) and Melospiza melodia (song sparrow).

A summary of results for pre-operational samples taken in 1980, which were separated into muscle, liver, bone and gut parts, is given in Table 33. Cs-137 and Co-60 were the only artificial radionuclides measured, with the exception of the herring gull liver sample which contained Mn-54. Highest activities were measured in starlings (up to 18

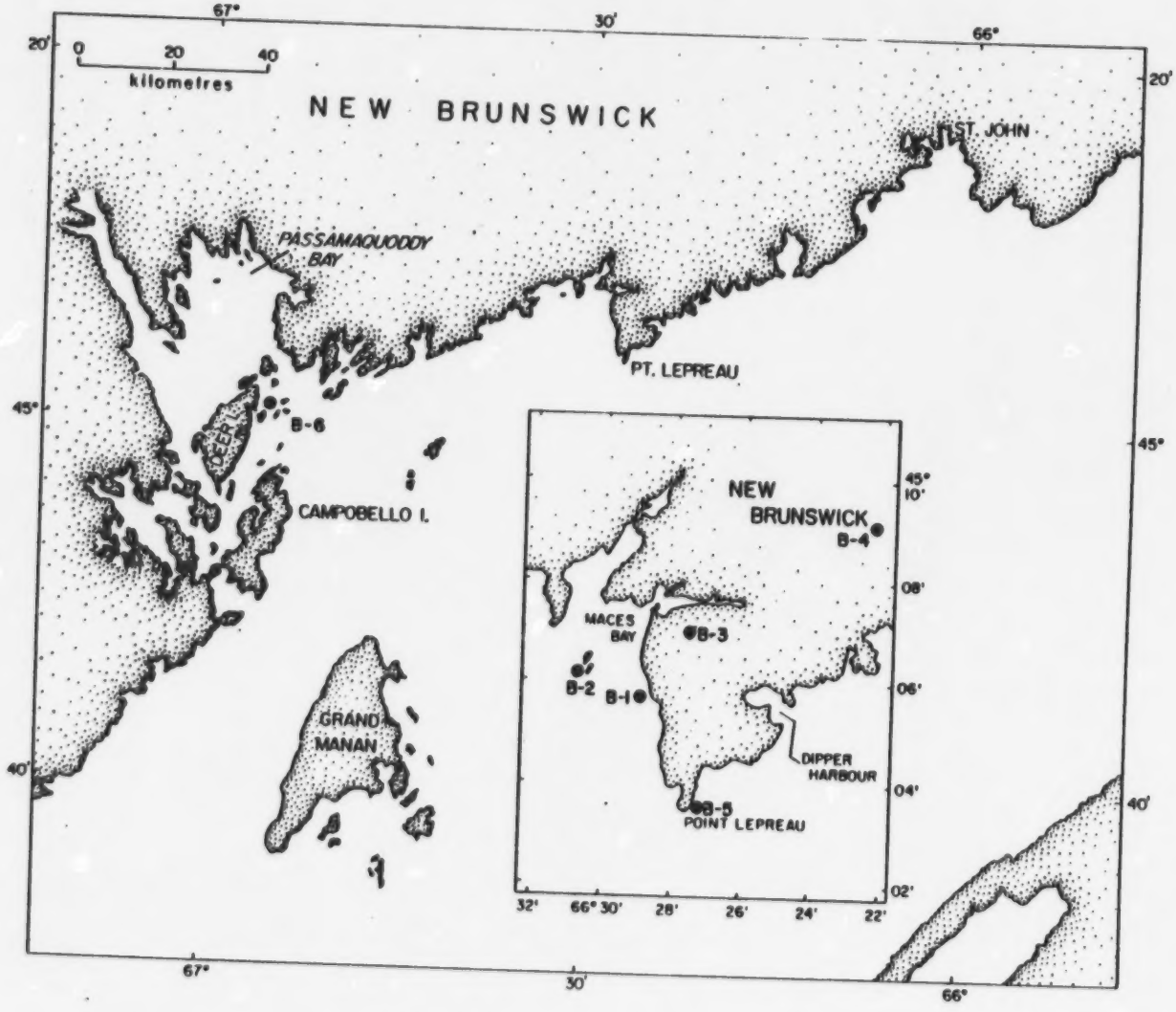


Figure 39. Sampling sites for birds collected in the vicinity of Point Lepreau.

Table 33. Average radionuclide results for birds collected near Point Lepreau.

Species	Organ	Collection Site	Collection Date	Sample Weight (g)	Sample Geometry	%H_2O	Cs-137 (Bq/kg)	Co-60 (Bq/kg)
Starling: (Sturnus vulgaris)	muscle	B-4	16/2/80	13.3	3	72.6	18 ± 10	18 ± 12
	liver	B-4	16/2/80	2.02	5	77.7	8.7 ± 4.3	10 ± 7
	bone	B-4	16/2/80	11.4	4	60.6	<2.8	<4.1
Song Sparrow (Melospiza melodia)	total	B-6	12/2/80	11.8	3	54.3	3.4 ± 1.8	3 ± 2
Eider (Somateria mollisima)	muscle	B-1	9/2/80	44.4	2	71.0	<2.0	1.5 ± 1
	liver	B-1	9/2/80	15.5	4	61.7	<4.5	<6.0
	bone	B-1	9/2/80	36.3	3	38.9	<2.6	<3.8
	gut	B-1	9/2/80	81.1	2	70.9	<1.5	0.8 ± 0.5
Cormorant (Phalacrocorax carbo)	muscle	B-2/B-1	9/2/80	54.2	2	69.4	1.5 ± 0.7	1.5 ± 1.0
	liver	B-2/B-1	9/2/80	24.5	3	68.6	1.7 ± 1.0	1.8 ± 1.3
	bone	B-2/B-1	9/2/80	19.0	4	39.6	<2.6	<6.0
	gut	B-2/B-1	9/2/80	144.3	1	57.8	0.5 ± 0.3	0.8 ± 0.5
Herring Gull (Larus argentatus)	muscle	B-3	16/2/80	23.5	3	65.5	1.7 ± 1.0	1.3 ± 0.7*
	liver	B-3	16/2/80	28.4	3	56.6	<2.3	<3.2
	bone	B-3	16/2/80	15.1	4	38.7	<3.6	4 ± 2
	gut	B-3	16/2/80	21.2	3	70.4	1.0 ± 0.8	<4
Purple Sandpiper (Calidris maritima)	muscle	B-5	9/2/80	11.4	4	74.0	<5.0	<8.2
	liver	B-5	9/2/80	2.8	5	77.4	<3.9	9 ± 5
	bone	B-5	9/2/80	5.0	5	63.3	<7	<10
	gut	B-5	9/2/80	11.3	5	65.6	<3.5	<5.2

* Other Radionuclides present
Mn - 54 - 4.0 ± 1.5 Bq/kg

Bq/kg dry) and song sparrows (average Cs-137 activity of 3-4 Bq/kg) with lowest activities measured in herring gulls, cormorants, eider ducks and sandpipers (Table 33). The higher levels found in the terrestrial birds reflect the higher levels of radioactivity present in their food (terrestrial plants and insects) compared to levels found in the food of the marine birds. The slightly higher levels (up to 1.8 Bq/kg) found in the herring gull compared to other marine birds probably reflects its mixed marine-terrestrial diet. The activities in the guts were found to be between 0.4 to 1.0 Bq/kg for Cs-137 and 0.8 Bq/kg for Co-60 in marine birds and 4.0 Bq/kg (Cs-137) and 2.3 Bq/kg (Co-60) for terrestrial birds, results which are in accord with the differences in radionuclide levels in their respective foods. Activities measured in marine birds are similar to those measured in marine vertebrates, suggesting that they feed on similar foods and accumulate radionuclides in a manner similar to marine vertebrates. Activities measured in the terrestrial birds agree well with levels measured by Willard (1960) for birds in an uncontaminated environment. Activities are generally an order of magnitude higher than those in marine mammals and slightly lower than those in other terrestrial and aquatic vertebrates, as was also observed by Davis (1963). This probably reflects differences in habitat, feeding habits and food.

7.0 LABORATORY INTERCOMPARISION STUDY

The AERU Laboratory maintains a comprehensive quality assurance program, run in two parts. Internally, the use of standards and blanks matched for each geometry and matrix used, ensures a routine check on all analytical results. Externally, the AERU laboratory participates in an intercomparison program managed by the Environmental Protection Agency (EPA, Las Vegas, Nevada) to maintain a check on the analytical precision and accuracy of the laboratory's radionuclide results. Laboratories throughout Canada and the United States participate in this program, which includes a wide range of radionuclides in various matricies. The results of the program are published and distributed to participants throughout the United States by the EPA.

The AERU results for 1979, 1980 and 1981 have been published previously (Smith et al., 1981, Smith et al., 1982) and have shown that no significant problems exist with the experimental procedures in operation. The results for 1981-83 are shown in Table 34 and include analyses of air filters for Cs-137, water for gamma-emitting radionuclides, Sr-90 and tritium and sediment for Ra-226 and its daughter products. Values determined by AERU are generally in good agreement with EPA values. Tritium in water and Cs-137 in air filters have consistently agreed with EPA values. Levels of I-131 in water are in good agreement with EPA values where postal delays have not been too great. An initial calibration error for Zn-65 in water has been corrected and since then Zn-65 determinations have been acceptable. A calibration error for Ra-226 in sediment, which resulted in a low result, was identified and corrected. Results for Sr-90 in water are included for the first time for 1982-1983 and show that the accuracy and precision for the techniques used in the Sr-90 analysis are very good.

Table 34. Results of AERU laboratory intercomparison experiments with the Environmental Protection Agency (Las Vegas, Nevada). Asterisk denotes results not included in EPA report.

Sample Matrix	Collection Date	Nuclide	EPA Value	AERU Value	Grand Mean (All Labs)	2 SD (AERU Expt.)	2 SD (All Labs)	AERU/ EPA
Water	13/2/81	H-3	1760 pCi/l	1760 pCi/l	1778 pCi/l	20	460	1.00
Water	12/2/81	H-3	1950 pCi/l	1930 pCi/l	1948 pCi/l	40	484	0.99
Water	9/10/81	H-3	2210 pCi/l	2277 pCi/l	2133 pCi/l	42	428	1.03
Water	11/12/81	H-3	2700 PCi/l	2630 pCi/l	2676 pCi/l	124	448	0.97
*Air Filter	19/12/80	Cs-137	19 pCi	19.7 pCi/filter	24 pCi	-	12	1.04
Air Filter	24/3/81	Cs-137	14 pCi	14.9 pCi/filter	16 pCi	4	8	1.06
Air Filter	25/7/81	Cs-137	19 pCi	20.5 pCi/filter	24 pCi	2	12	1.08
Air Filter	25/7/81	Cs-137	19 pCi	20.4 pCi/filter	24 pCi	-	12	1.07
Air Filter	26/3/82	Cs-137	23 pCi	23.3 pCi/filter	25 pCi	2	14	1.01
Water	6/2/81	Co-60	25 pCi/l	25 pCi/l	25 pCi/l	2	8	1.00
Water	6/2/81	Zn-65	85 pCi/l	47 pCi/l	89 pCi/l	10	22	.53
Water	6/2/81	Cs-134	36 pCi/l	33 pCi/l	33 pCi/l	6	10	.92
*Water	6/2/81	Cs-137	4 pCi/l	4 pCi/l	5 pCi/l	-	4	1.00
*Water	9/10/81	Cs-137	15 pCi/l	16 pCi/l	16 pCi/l	-	6	1.07
*Water	4/12/81	I-131	76 pCi/l	75.3 pCi/l	69 pCi/l	-	20	1.01
Water	5/2/81	Cs-137	23 pCi/l	18.3 pCi/l	24 pCi/l	2	8	1.25
Water	2/4/82	I-131	62 pCi/l	52 pCi/l	63 pCi/l	12	16	1.18
Water	11/6/82	H-3	1830 pCi/l	1823 pCi/l	1765 pCi/l	50	458	0.996
*Water	8/10/82	H-3	2560 pCi/l	2546 pCi/l	2517 pCi/l	80	500	0.995
Water	10/12/82	H-3	1990 pCi/l	1976 pCi/l	2009 pCi/l	128	466	0.993
Water	4/6/82	Cs-137	25 pCi/l	25 pCi/l	27 pCi/l	6	8	1.00
		Cs-134	35 pCi/l	30 pCi/l	34 pCi/l	4	8	0.86
		Co-60	29 pCi/l	27 pCi/l	31 pCi/l	6	8	0.93
		Zn-65	26 pCi/l	23 pCi/l	27 pCi/l	12	12	0.88

Table 34. Cont'd.

Sample Matrix	Collection Date	Nuclide	EPA Value	AERU Value	Grand Mean (All Labs)	2 SD (AERU Expt.)	2 SD (All Labs)	AERU/ EPA
KCFC	5/02/82	Cs-137	23	18	24	2	8	0.78
		Cs-134	22	11	21	2	6	0.50
Air Filters	26/03/82	Cs-137	23 pCi/Filter	23.3 pCi/Filter	27 pCi/Filter	2	12	1.01
Air Filters	24/09/82	Cs-137	27 pCi/Filter	22 pCi/Filter	25 pCi/Filter	0	8	0.82
Air Filters	26/11/82	Cs-137	27 pCi/Filter	28 pCi/Filter	30 pCi/Filter	2	10	1.04
Other	24/09/82	Pb-210*	5.2	5.3	4.4	0.6	5.4	1.00
		Ra-226	5.2	2.7	4.3	0.1	4	0.52
		Po-210	5.2	5.3	-	0.6	-	1.00
Water	11/02/83	H-3	2560 pCi/1	2538 pCi/1	2534 pCi/1	92	273	0.99
Water	10/06/83	H-3	1529 pCi/1	1550 pCi/1	1552 pCi/1	10	202	1.01
Water	1/04/83	I-131	26.8 pCi/1	26 pCi/1	26.5 pCi/1	4.0	9.2	0.97
Water	7/01/83	Sr-90	17.2 pCi/1	18.3 pCi/1	16.7 pCi/1	1.6	3.4	1.06
Water	9/05/83	Sr-90	13 pCi/1	10 pCi/1	13 pCi/1	4	4	0.77
		Co-60	30 pCi/1	28 pCi/1	31 pCi/1	9	8	0.93
		Cs-134	33 pCi/1	25 pCi/1	31 pCi/1	6	8	0.76
		Cs-137	27 pCi/1	26 pCi/1	27 pCi/1	6	8	0.96
Water	6/05/83	Sr-90	37.7 pCi/1	39 pCi/1	37.1 pCi/1	0.4	9.2	1.03
Water	3/06/83	Co-60	13 pCi/1	16 pCi/1	14 pCi/1	2.2	4	1.23
		Zn-65	36 pCi/1	34 pCi/1	37 pCi/1	3.2	12	0.94
		Ru-106	40 pCi/1	39 pCi/1	40 pCi/1	33	14	0.98
		Cs-134	47 pCi/1	41 pCi/1	44 pCi/1	1	8	0.87
		Cs-137	26 pCi/1	27 pCi/1	28 pCi/1	4.5	10	1.04

Postal delays resulted in some of the AERU results (denoted by an asterisk) not being included in the EPA reports. Results indicate that there are no apparent precision or accuracy problem with AERU results for the analysis of radionuclides distributed in this program.

8.0 CONCLUSIONS

(1) The measurement of pre-operational levels of radionuclides has been completed for a wide range of environmental phases. These levels have been used to provide insight into transport pathways and mechanisms which will govern the distribution of radioactivity released from the Point Lepreau NGS.

(2) The results for the drifter return program indicate that dissolved and thermal contamination released from the reactor will follow water circulation of the Bay of Fundy and that particle-reactive releases will likely be deposited in the Maces Bay - Passamaquoddy region in the summer and fall and may be redistributed by winter and spring storms.

(3) Levels of tritium in the Bay of Fundy range from <1.5 Bq/l to 3.4 Bq/l ($\bar{x}$ = 2.2 Bq/l) and exhibit no seasonal or geographical anomalies. Less than 10% of total Cs-137 was measured on particles. Cs-137 activities range from 2.5 mBq/l to 6.0 mBq/l with an average value of 3.5 mBq/l. No strong correlation between Cs-137 and other parameters, such as depth, salinity or geographical location, was observed, although there is some evidence that Cs-137 is desorbed from freshwater particulate material in the Bay of Fundy during periods of high river discharge. Strontium-90 activities range from 1.3 mBq/l to 4.8 mBq/l with an average value of 3.0 mBq/l. The mean measured Cs-137/Sr-90 activity ratio of 1.4 is closed to the fallout ratio of 1.5.

(4) Levels of total Cs-137 in freshwater range from <1.2 mBq/l to 4.7 mBq/l ($\bar{x}$ = 2.7 mq/l), with no differences between stream and lake sites. Particulate Cs-137 contributes 46% of the total and an

average distribution coefficient (between dissolved and particulate) of approximately 8 x 10^5 Bq/g/Bq/ml was measured. An average tritium activity of 6.0 mBq/l (range 4.2 mBq/l to 10.6 mBq/l) was measured during the pre-operational phase.

(5) Sediment inventories of Pb-210, Cs-137 and plutonium isotopes measured in the Bay of Fundy have been used to identify the immediate vicinity of Point Lepreau as an area of minimum deposition of fine-grain material and the Quoddy region, near the approaches to Passamaquoddy Bay, as an area of deposition of fine-grain material which may become an active future site for the accumulation of reactor-released radionuclides.

(6) The record of fallout radionuclide fluxes is preserved better in aquatic sediments than in more easily eroded soils, although details of the former record are obscured in many cases by sediment mixing due to biological activity.

(7) Levels of tritium in atmospheric moisture exhibit no significant seasonal or yearly bias. An average level of 5.8 Bq/l of air moisture (standard deviation = 2.8 Bq/l,n=72) was measured during the pre-operational phase. Levels of fallout radionuclides in air particulate material in 1982 were below the detection limit. Fallout products such as Ce-141, Ce-144 and Zr-95, produced during the 1980 Chinese nuclear test and measured on air particulate matter in 1981, were used to determine the source and timing of these radionuclide inputs into the environment.

(8) Pre-operational levels of fission products in marine organisms are very low and in agreement with literature results. Cesium-137 was the most commonly observed fission product, Co-60 was measured in

several samples and Zr-95 and Mn-54 were measured in several algae samples collected the summer following the 1980 Chinese nuclear test.

(9) Baseline radionuclide levels were several orders of magnitude greater in terrestrial samples than those measured in marine organisms. Short-lived fallout radionuclides such as Ru-103 and Zr-95, produced during the 1980 Chinese nuclear test were observed in all plants sampled in the summer following the test. Levels of fallout radionuclides were two to ten times greater in lichen and moss compared to other plants. Blueberry and horsetail plants were the most effective of the remaining plants in accumulating fallout products.

(10) The lower levels of radionuclides in marine phases compared to terrestrial phases are due both to the presence of a large diluting seawater reservoir and the high ionic strength of seawater which acts to limit the sorption of radionuclides onto particle surfaces.

(11) Lichen are particularly efficient accumulators of air particulate material and serve as a good indicator species for the time-integrated inventory of radioactivity in the atmosphere. Lichen are cost-effective air particulate samplers and the radioactivity determinations have an excellent signal to noise ratio.

(12) The highest fallout radionuclide levels were measured in the organs of birds having a terrestrial diet while reduced levels were found in birds which tend to feed on marine organisms, a result which is consistent with the generally higher levels of radioactivity measured in terrestrial environmental phases.

(13) The average tritium levels measured in marine organisms ($\bar{x}$ = 2.6

(13) Bq/l, standard dev. = 0.4, n = 13) and terrestrial organisms ($\bar{x}$ = 4.9 Bq/l, standard dev. = 1.8, n = 18) are within the range of tritium concentrations measured in seawater (<1.5 Bq/l - 3.4 Bq/l) and freshwater (4.2 Bq/l to 10.6 Bq/l) respectively and indicate that tritium is associated with the hydrological cycle. One unusual observation was made for shell material of marine mollusc and crustacea where average levels (5.3 Bq/l, standard dev. = 1.0, n = 3) where higher than those measured in seawater indicating some process of accumulation of tritium in shell material.

(14) There is no evidence, during the pre-operational phase of the monitoring program, to suggest the concentration of fallout radionuclides with increasing trophic level.

ACKNOWLEDGEMENTS

The working group on Point Lepreau Environmental Monitoring is composed of scientists from different government departments, (Appendix 1) all of whom have contributed to the development of this program. In particular, we wish to thank Gary Bugden (Bedford Institute of Oceanography) who supervised the physical oceanographic component of this program, E. Brandon and B. Taylor (Atmospheric Environmental Service, Bedford, N.S.) who supervised the meteorological aspects of PLEMP and Dr. P. Yeats and B. Amirault (Bedford Institute of Oceanography) for their analysis and interpretation of trace metal results. Equally helpful have been Dr. D. Scarratt (Fisheries Biological Station, St. Andrew's, N.B.) who has guided much of the marine biological work; K. Brown (Environment New Brunswick, Fredericton, N.B.) who supervised much of the terrestrial and aquatic sampling, and B. Johnson (Canadian Wildlife Service, Sackville, N.B.) who supervised the bird sampling. In addition, Mia Huh, J. Abriel, Marie Svenson, Dr. David Assinder and Jeff Bishop have provided substantial analytical support in the AERU laboratory analysing samples for Pb-210, Cs-137, Pu and H-3 and Dr. J.M. Bewers, (Bedford Institute of Oceanography) has provided continual support and many helpful discussions throughout all stages of this program.

Residents of Point Lepreau, N.B. who have provided great assistance in the operation of the air monitoring stations include Mr. and Mrs. G. Thompson (Dipper Harbour), Mrs. J. Pollock (Musquash) and Mr. R. Mawhinney (Welch Cove) while Mr. M. Durkee has kindly provided similar support in the operation of an air monitoring station at Digby, N.S. Appreciation is also extended to those members of the N.B. Electric Power Commission's Health Physics Department who helped with the deployment and operation of air

monitoring station equipment, notabley Dr. J. Paciga, Mr. C. Nasin and Mr. J.B. McCulley.

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Appendix 1. Conversion factors for different radioactivity units.

1 Becquerel (Bq)	= 2.7×10^{-11} Curies (Ci)
1 Bq	= 27 pCi
1 Bq	= 60 dpm
1 Bq/l	= 8.3 Tritium units (TU)
1 Ci	= 3.7×10^{10} Bq
1 mCi	= 3.7×10^{7} Bq
1 pCi	= 3.7×10^{-2} Bq
1 dpm	= 0.0167 Bq
1 dpm	= 2.2 pCi
1 TU	= 0.12 Bq/l
1 TU	= 7.2 dpm/l
1 TU	= 3.2 pCi/l

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Nuclide	Half-life	Geometry 1	Geometry 2	Geometry 3	Geometry 4	Geometry 5
Ag-110 m	253 d	150	94	95	95	47
Ba-140	12.8 d	1700	1700	1000	1000	490
Be-7	53.6 d	1100	1200	690	670	370
Ce-141	32.4 d	280	330	170	140	110
Ce-144	284 d	780	900	460	400	200
Co-56	77 d	130	140	90	90	40
Co-57	270 d	110	130	63	54	35
Co-58	71 d	130	140	77	78	39
Co-60	5.26 y	120	120	65	69	35
Cr-51	28 d	1700	1800	1100	1001	570
Cs-134	2.07 y	110	120	69	69	33
Cs-137	30.2 y	107	100	62	60	31
Eu-152	12.5 y	660	690	410	410	200
Eu-155	1.81 y	300	340	160	160	69
Fe-59	44.6 d	340	330	190	200	89
I-131	8.06 d	1200	1400	780	750	410
La-140	40.2 h	510	510	340	360	190
Mn-54	313 d	110	110	67	65	31
Nb-95	35.1 d	180	180	110	110	54
Ru-103	39.6 d	150	160	93	92	46
Ru-106	1.0 y	900	940	540	550	250
Sb-124	60.2 d	18	20	11	11	6
Sb-125	2.7 y	240	260	150	147	73
Zn-65	244 d	290	260	140	150	73
Zr-95	64 d	240	240	150	150	70

Appendix 3. Detection limits (Bq/kg) for each radionuclide and geometry assuming a counting time of 10^5 sec for analyses performed one month subsequent to sample collection. Detection limit for specific sample is determined by choosing detection limit for a radionuclide for the appropriate geometry and dividing by the sample weight. These estimates are approximate and will vary depending upon the specific counting times and delay between sample collection and analysis.

Nuclide	Half-life	Geometry 1	Geometry 2	Geometry 3	Geometry 4	Geometry 5
Ag-110 m	253 d	380	410	240	240	120
Ba-140	12.8 d	-	-	-	-	-
Be-7	53.6 d	-	-	-	-	-
Ce-141	32.4 d	-	-	-	-	-
Ce-144	284 d	1700	2000	1000	890	660
Co-56	77 d	2600	2800	1600	1600	750
Co-57	270 d	250	300	150	140	82
Co-58	71 d	3400	3400	1900	1900	980
Co-60	5.26 y	130	130	74	78	40
Cr-51	28 d	-	-	-	-	-
Cs-134	2.07 y	150	160	93	940	45
Cs-137	30.2 y	110	110	63	61	32
Eu-152	12.5 y	700	720	440	430	200
Eu-155	1.81 y	420	480	220	230	98
Fe-59	44.6 d	57000	56000	32000	30000	15000
I-131	8.06 d	-	-	-	-	-
La-140	40.2 h	-	-	-	-	-
Mn-54	313 d	230	230	140	135	65
Nb-95	35.1 d	-	-	-	-	-
Ru-103	39.6 d	49000	53000	31000	30000	15000
Ru-106	1.0 y	1700	1800	1000	1000	470
Sb-124	60.2 d	800	880	500	490	250
Sb-125	2.7 y	300	330	180	180	91
Zn-65	244 d	730	680	370	383	190
Zr-95	64 d	7900	8200	4900	4900	2400

Appendix 4. Detection limits (Bq/kg) for each radionuclide and geometry assuming a counting time of 10^5 sec for analyses performed one year subsequent to sample collection.